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Municipal Solid Waste Generated Gas and Leachate: 1974 Summary Report

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16. ABSTRACT

The report reviews literature and 1974 research dealing with the degradation of municipal solid waste in sanitary landfills, production of leachate and gas, the modes of migration of these products, their environmental effects, and methods of control. Leachate production can be signifiantly affected by selection and design of landfill cover material. Whenleachate is formed the per capita discharge of water-borne contamininants over a long period will be of the same order of magnitude as per capita discharges into a sanitary sewerage system. Soils, especially those that are fine-grained and of low permeability, are very effective in controlling the migration of leachate. Biological, physical-chemical, and land-based wastewater treatment methods are effective when leachate must be treated.

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November 1984

A 1974 SUMMARY REPORT: MUNICIPAL SOLID WASTE GENERATED GAS AND LEACHATE

MUNICIPAL ENVIRONMENTAL RESEARCH LABORATORY
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FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplies, and to minimize the adverse economic, social, health,and aesthetic effects of pollution. This publication is one of the products of that research; a most vital communications link between the researcher and the user community.

This report summarizes and evaluates information on the production of leachate and landfill gas in sanitary landfills, methods for controlling these byproducts of biological activity and the potential adverse effects of uncontrolled migration.

Francis T. Mayo, Director
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ABSTRACT

A briefing document was prepared in 1974 for the Office of Solid Waste Management programs, (now the Office of Solid Waste) USEPA. The purpose of the document was to describe then current knowledge of the degradation of landfilled municipal solid waste, production of leachate and gas, migration of these products, their effects, and methods for control. This report is an edited, but not an updated version of that briefing document.

The report reviews literature and current research dealing with the degradation of municipal solid waste in sanitary landfills, production of leachate and gas, the modes of migration of these products, their environmental effects, and methods of control. Leachate production can be significantly affected by selection and design of landfill cover material. When leachate is formed the per capita discharge of waterborne contaminants over a long period will be of the same order of magnitude as per capita discharges into a sanitary sewerage system. Soils, especially those that are fine-grained and of low permeability, are very effective in controlling the migration of leachate. Biological, physical-chemical, and land-based wastewater treatment methods are effective when leachate must be treated. Additional research is required to allow prediction of the movement of contaminants through soils, and to provide design criteria for cover materials, to evaluate treatment on a field scale, to determine optimum rates of solid waste stabilization, and to assess the presence and significance of pathogens in soils.

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SECTION 1

INTRODUCTION AND SUMMARY

The Office of Solid Waste Management Programs in 1973 requested the Solid and Hazardous Waste Research Laboratory to prepare a briefing document on Landfilled Municipal Waste. The report was to summarize information on municipal refuse decomposition, leachate and gas production, and the effects and control of these biological products. The briefing report was distributed in limited quantity at a conference held in Washington, D.C. during August, 1974.

The briefing report was edited for publication in 1979 as a background document for those unfamiliar with the problems of solid waste disposal. Advances have been made in several technical areas, since the report was originally prepared; notable, in the areas of polymeric liners and movement of leachate through the soil. The reader is refered to the follwoing list of reports for more recent information:

- Gentelli, Emil J. and John Cirello. Gas and Leachate from Landfills. Proceedings of a Research Symposium held at Rutgers University, New Brunswick, New Jersey, March 1975. EPA-600/9-76-004. U.S. Environmental Protection Agency, Cincinnati, Ohio. 1976.
- Banerji, S.K. Management of Gas and Leachate in Landfills. Proceedings of the Third Annual Municipal Solid Waste Research Symposium held at St. Louis, Missouri, March 1977. EPA-600/9-77-026. U.S. Environmental Protection Agency, Cincinnati, Ohio. 1977.
- Fuller, W.H. Residual Management by Land Disposal, Proceedings of the Hazardous Waste Research Symposium held at the University of Arizona, Tucson, Arizona, February 1976. EPA-600/9-76-015. U.S. Environmental Protection Agency, Cincinnati, Ohio. 1976.
- Shultz, D.W. Land Disposal of Hazardous Wastes. Proceedings of the Fourth Annual Research Symposium held at San Antanio, Texas, March 1978. EPA-600/9-78-016. U.S. Environmental Protection Agency, Cincinnati, Ohio. 1978.

The sanitary landfill method has been advocated for the past 3 decades for disposal of high volumes of municipal solid waste. These wastes consist of mildly infectious, readily decomposable organics and other organic and inorganic materials. The sanitary landfill method was developed because such waste materials represent a threat to public health[1,2] and to environmental quality. When the landfill is properly designed and operated, it can be a means of reclaiming marginal land while achieving the primary objective of economical, healthful, and environmentally safe solid waste disposal.

Unfortunately, large volumes of municipal solid waste often are disposed of simply by dumping or by open burning rather than by incorporation into an environmentally sound sanitary landfill. Improper site selection, and improper design, operation, and maintenance of such dumps (often spoken of as "landfills") have contributed to the leaching of organic and inorganic constituents as water percolates through the disposed waste. These materials can reach streams, underground aquifers, lakes, and impoundments[3-7] and thus can seriously impair water quality and endanger the health and welfare of communities nearby.

Even in a well-operated sanitary landfill various physical, chemical, and biological processes invariably produce compounds that are susceptible to being dissolved or suspended in water as it percolates through the solid waste. The volume of leachate produced at any site depends on many factors, as does its composition.

Decomposing solid wastes also produce gases, which like the leachates can be hazardous. Uncontrolled accumulations of landfill gas can be explosive and are often malodorous. Again, the composition of the gases and the volumes produced are governed by a variety of factors specific to each landfill site. Methods of control likewise must be tailored to a highly specific set of local circumstances. Moreover, under certain conditions it is feasible to control the adverse effects of landfill gas not by limiting its production but rather by optimizing its production for use by nearby consumers.

This report deals with the practical aspects of landfill-generated leachates and gases, presenting quantitative information that might be applied to the siting, design, and operation of a sanitary landfill. It also provides supporting theoretical discussions and recommends promising areas for additional study.

Section 2 describes the composition of municipal solid waste and the mechanisms by which it is biologically degraded in a sanitary landfill. Section 3 summarizes the methods of sampling and analysis. Sections 4, 5, and 6 concern leachate:

how it is produced and what it is composed of; how it migrates, and the resultant environmental and health effects; and methods of leachate control. The three remaining sections constitute a similar analysis of landfill gas. The following paragraphs summarize some of the salient features of the report and present recommendations for further research.

SUMMARY

Municipal Solid Waste Degradation

Leachate and gas production problems are related to the composition of waste materials, the operational procedures, the site's environment (temperature, hydrogeology, etc.), and the biological activity within the solid waste. The moisture and temperature regimes of landfilled solid waste strongly affect the survival of viruses and fecal organisms. On the basis of experience with septic tank leaching systems and cesspools, extensive migration of pathogens and fecal indicator organisms as leachate moves through soils is not expected.

Analytical Methods

Methods used to analyze highly contaminated aqueous samples are generally applicable to leachate analysis. Unless extensive dilution is practical, colorimetric techniques should not be used because of the color intensity of leachate. Use of the membrane filter technique for bacteriological assay has yielded values 100 times lower than those obtained by the most probable number (MPN) multiple-tube technique. The sample collection, preservation, and storage techniques significantly affect analytical results.

Leachate Production

The volume of leachate produced at a sanitary landfill site varies widely and is determined by the water balance. Adding moisture to solid waste while it is being deposited may cause the early appearance of leachate but will not significantly affect the long-term volume of leachate produced. However, the effect on decomposition may be significant.

Leachate derived from recently deposited municipal solid waste is generally acidic (pH 5.0 ± 1.0) and high in biologically degradable organics (BOD ≅ 10,000 mg/liter, COD ≅ 16,000 mg/liter, and TOC ≅ 5,700 mg/liter) and total solids (> 1 percent). It also contains indicators of fecal contamination. Leachate derived from solid waste that has undergone extensive degradation generally is more alkaline (pH ∿ 7) and is low in biologically oxidizable organics (BOD < 100 mg/liter, COD ≅ 1,000 mg/liter, TOC ≅ 300 mg/liter) and total solids (2,000 mg/liter). Concentrations of heavy metals in leachate are generally less than 1 mg/liter; zinc concentrations, however, can range from 10 to 100 mg/liter, and lead concentrations occasionally exceed 1 mg/liter. Iron is commonly present at 1000 mg/liter.

Municipal solid waste exposed to 660 mm (26 in) of annual infiltration was found to yield 42.6 g (0.1 lb) COD, 25.3 g (0.06 lb) total solids, and 23.0 g (0.05 lb) BOD_5 per capita per day. The per capita discharge of waterborne contaminants (e.g. COD) from landfills (prior to reaction with soils) are of the same order of magnitude as wastewater discharges to sewage systems.

Leachate Migration and Effects

The fate of contaminants carried in leachate is affected by physical processes, such as filtration, dispersion, and mixing, and by soil chemical processes, such as ion exchange, adsorption, and oxidation and reduction. Most of the suspended matter and microorganisms are removed by filtration through a few meters of soil. Materials not filtered compete with other contaminants for adsorption sites.

Dispersion cannot be relied on as a means of contaminant removal. Dispersion is negligible in a narrow range of grain size but increases as soil materials become less uniform.

Because of their large cation exchange capacity (CEC) and surface area, the clay minerals (e.g. montmorillonite, illite kaolinite, chlorite and vermiculite) are the most important active fraction of soils in attenuating contaminants carried in leachate. The efficacy of attenuation varies among the clay mineral groups. The microbial flora and the extent of aeration in the soils through which leachate migrates can profoundly affect its composition.

In general, the attenuation of leachate is not permanent because the adsorption and ion exchange mechanisms are partially reversible. Leachate contamination of surface streams and ponds has led to extensive biological growths that adversely affect the distribution of aquatic species.

Leachate Control Methods

The behavior of leachate from landfills is affected by measures taken during the design and construction phases as well as during operation of the fill. These include selection of the landfill site on the basis of soil textures, mineralogy, and groundwater flow patterns; blending of coarse and fine soils during construction; baling or compaction of the waste fill materials; control of the slope and density of cover materials; maintenance of grades in accordance with design contours; control of vegetation; and construction of subsurface drains and ditches to control groundwater movement. These and similar control measures are based on preventing the infiltration of water and thus limiting the generation of leachate.

In contrast, production of leachate may be maximized by encouraging the infiltration of water to achieve rapid biological stabilization of waste materials. Rapid stabilization reduces the period over which the fill must be operated and maintained and allows earlier conversion of the landfill site to other uses.

The greatest control is achieved by using liners to collect the generated leachate for subsequent treatment. Native soil, soil admixtures, and synthetic liners are being evaluated for leachate collection. Thin membranes, such as those used in bituminous spray seals, and polymeric membranes, must be protected with layers of soil to prevent punctures and tears. Methods of installing and sealing the liners are being evaluated as well as the liner materials. In an experimental field study, greater than 99 percent of the leachate generated in a landfill was collected from a clay liner that had a hydraulic conductivity of approximately 10^{-5} cm/sec (3.9×10^{-6} in/sec).

Methods of treating the collected leachate are under study. Biological treatment methods reduce the level of contaminants in leachate derived from actively decomposing solid waste, but physical-chemical methods are generally ineffective. The reverse is true of leachates that are derived from stabilized solid waste or contain large quantities of suspended matter; these are more effectively treated by physical-chemical than by biological methods.

When leachate emanating from actively decomposing solid waste is treated aerobically, the residual COD is generally less than 500 mg/liter and some polyvalent cations are reduced significantly (manganese by 23 percent and iron by 96 percent). Nutrient addition usually is not required. When actively decomposing solid waste is treated anaerobically, either in a digester or by being passed through an anaerobic filter, the residual COD is usually less than 200 mg/liter and some polyvalent cations are reduced significantly (zinc by 31 percent and manganese by as much as 98 percent). Lime treatment removes organic matter with molecular weight greater than 50,000, and removes substantial quantities of metals. An activated carbon column should be operated as an upflow fluidized bed if used to treat leachates high in BOD; otherwise a packed-bed, downflow column is more economical. Treatment with ozone is less effective than with activated carbon. Only organically stabilized leachates are effectively treated by reverse osmosis.

Addition of leachate to sewage should not exceed 5 percent by volume because addition in larger amounts impairs settling characteristics. Spraying leachate on land is effective in removing organic and inorganic contaminants but may lead to high levels of nitrate (up to 50 mg/liter) if application is not optimized to allow denitrification. Recirculating leachate through decomposing solid waste enhances the rate of stabilization, which can be accelerated under controlled pH conditions. All methods of leachate treatment produce a residual that must be disposed of.

Gas Production

The composition and rates of gas production depend on the specific environmental conditions that affect biological activity, moisture and temperature being the most important.

Solid waste containing 75 percent volatile matter theoretically can produce 0.63 m^3/kg (10.1 ft^3/lb) (dry) of gas under anaerobic conditions. Maximum gas production measured in a completely mixed reactor at 60°C (140°F) was 0.45 m^3/kg (7.2 ft^3/lb) of volatile solids added. At 40°C (104°F), gas production was only 0.37 m^3/kg (6 ft^3/lb). The mechanisms and the temperature and pH conditions that enhance methane fermentation in landfilled solid waste are similar to those for digestion of sewage sludge. Methane fermentation proceeds best at solid waste moisture contents greater than 100 percent (dry weight).

The only gases produced in large volumes by degrading solid wastes are carbon dioxide and methane. Methane is produced only under anaerobic conditions.

Gas Migration and Effects

Uncontrolled migration of landfill-generated gas have caused death, forced evacuation of buildings, temporarily increased concentrations of dissolved solids in local groundwaters, and stunted or killed vegetation.

Landfill gases generally diffuse through the soil because of partial pressure gradients, though pressure flow may also be induced because of saturated or frozen soils or impervious covers on the landfill. Carbon dioxide that migrates from a landfill may dissolve in an aquifer and form carbonic acid; calcium, magnesium, or other soluble material may be released when the buffering capacity of the aquifer is exhausted. Slightly more than 5 percent of the carbon dioxide produced at the rate of 560 tons/h per year and of methane produced at 56 tons/h per year remained in the ground; the remainder vented into the atmosphere. Field study indicated that methane comprised up to 5 percent of the gases in the soil as far as 137 m (452 ft) from the edge of the landfill.

Gas Control Methods

Composition of landfill gas and rates of production can be controlled to a limited extent by controlling the moisture regime. More positive control of gas movement is provided by barriers or vents. In addition to being almost impermeable to gases, barriers should be designed and constructed to accommodate differential settlement. Vents in which pumps maintain a negative pressure have been used to control gas and are particularly useful in recovering methane.

Construction materials may range from native soils (clay and gravel) to synthetic membranes and soil admixtures. Control of moisture content is important in maintaining the integrity of clay barriers and gravel trenches.

Recovery and utilization of landfill gas is technically and economically feasible under favorable conditions of gas generation, location, and demand. The potential uses are as a fuel and as a process feedstock. The time period over which supplies of landfill gas can be assured is critical to competitive marketing. Current estimates of gas recovery range from 0.013 to 0.047 standard m^3/kg (0.2 to 0.75 standard ft^3/lb).

RECOMMENDATIONS

Following are recommendations for research that would provide information necessary to predict sanitary landfill behavior and to evaluate the environmental impacts.

1. Innovative landfill disposal techniques should be developed on the basis of current understanding of the physical, chemical, and microbiological processes associated with solid waste stabilization.

2. Additional data gathering in field and laboratory studies is needed for determination of surface runoff/ infiltration rates under conditions prevalent at sanitary landfill sites. Such determination is essential in predicting leachate volume, assessing environmental impacts, and developing control technologies.

3. Studies should be done to assess the importance of the moisture regime in determining quantities and types of materials leached from decomposing solid waste. The feasibility of leachate recirculation and water management of large-scale facilities should be evaluated.

4. Assessment of the presence and survival of pathogens in leachate is needed for guidance with regard to leachate handling, treatment, and disposal.

5. The development of a quantitative model for predicting the attentuation of leachate in soils is not practical, but development of data and methods for qualitative prediction of the ability of various soils to attenuate the materials in solutions passing through them should be pursued virgorously.

6. Responsibility for effecting remedial measures and maintenance at closed sites should be clearly defined; associated legal, social, and economic factors must be studied to develop a feasible and effective arrangement that protects the environment.

7. Assess the hazard to the waste collection worker and to the public during storage, collection, transport, and disposal operations.

8. Appropriate loading rates to keep nitrate formation and soil clogging at a minimum need to be determined for land application of leachates.

9. Addition of industrial and municipal sludges and nutrients to the landfill should be studied to determine their effects on gas production.

10. Studies of gas migration and of controls to prevent undesirable migration are recommended.

11. Design criteria for constructing vents and barriers to prevent landfill gas migration are also needed.

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SECTION 2

WASTE COMPOSITION AND DEGRADATION

Problems associated with leachate and gas production are directly related to the composition of waste materials deposited at a landfill site as well as to the operational procedures and the geographic and hydrogeological conditions at the site. Regional changes in composition of the waste are attributable to climatic, economic, and demographic factors. It is therefore difficult to accurately characterize municipal solid wastes on a national basis, and then use such a characterization to determine the input to any particular landfill.

COMPOSITION

Solid wastes include a heterogeneous mixture of inorganic and organic constituents in a solid or semisolid form. The constituents may be highly reactive or they may be essentially inert and resistant to chemical, physical, or microbiological change. The degree or rate of change will depend upon opportunity for reaction; in this respect environmental factors and catalysis are extremely important. Many natural processes proceed readily within the environs of a landfill site.

As a basis for examining the processes that may cause production of leachate and gas, the data in Table 1 indicate the characteristics of municipal solid waste placed in Test Cell 1 at the EPA Field Site in Boone County, Kentucky, during the early summer of 1971. Potential problems with leachate and gas production are influenced by both inorganic and organic constituents of the waste. Leachate production relates mostly to the inert and organic fractions, and gas production to the decomposable or putrescible fractions, mainly food and garden wastes, paper, and related products. Other types of wastes are often disposed of in land disposal sites. Block[3] identified nonindustrial sources of toxic and hazardous wastes commonly disposed of in sanitary landfills, and EPA reported to Congress on industrial sources of hazardous waste streams.[4] Nonhazardous industrial and commercial wastes are also commonly disposed of on land.

TABLE 1. MUNICIPAL SOLID WASTE COMPOSITION

Constitutent	Composition of U.S. postconsumer net solid waste[a] % as disposed	Composition[b] % by wet weight	Moisture content % by dry weight	Moisture content % by wet weight
Food waste	13.3	4.3	106.0	51.9
Garden waste	14.1	9.3	131.0	55.9
Paper	39.6	50.9	57.5	36.3
Rubber, leather, plastics	6.8	7.8	25.8	20.6
Textiles	1.6	4.6	30.7	18.6
Wood	3.6	1.8	21.5	18.2
Metals	9.9	8.0	5.8	5.4
Glass and ceramics	10.3	6.4	1.3	0.6
Rock, dirt, ash	1.5	1.5	15.0	12.9
Fines (<25.4 mm)	c	2.9	39.7	28.8
Diapers	c	0.8	162.9	61.1
Total sample	100.00	98.3	40.5	27.6

[a] Reference 1. As-disposed basis assumes moisture transfer among materials in collection and storage but no net addition or loss of moisture for the aggregate of materials.

[b] Reference 2. Average of 12 random samples separated 1-2 weeks after collection.

[c] NA - Not available.

The moisture content of solid waste when deposited at a land disposal site is normally 15 to 35 percent of wet weight,[5] but it usually increases with the influx of rainfall or groundwater and with microbial activity. Moisture is critical for the occurrence of biological and many chemical reactions, and it is the medium that carries pollutants from the site as leachate.

FATE OF SOLID WASTE COMPONENTS DURING LANDFILL DISPOSAL

Little is known about characteristics of solid waste at most landfill sites except as analytically inferred when pollution problems arise. The data obtained in investigating gas and leachate problems are amenable to interpretations regarding the age and condition of the site, the degree and extent of reaction or stabilization of the solid waste constituents. Evaluation is complicated by many possibilities for oxidation-reduction, coagulation, precipitation, and acid-base and complexation reactions that may occur in solution, and by the opportunity for adsorption, ion-exchange, filtration, and other reactions within the solid waste mass or surrounding environment. Some of the principal reactions that determine pollution potential are discussed below.

MICROBIAL MEDIATION AND ATTENUATION

The greatest pollution problems occur when organic content of the leachate is high and when gas is produced in such quantities and forms that its solution causes groundwater problems or its release, migration, and subsequent accumulation create an explosion hazard. Both leachate and gas production can be traced to biochemical reactions occurring within the site. These reactions are microbially mediated and yield intermediate and end products that may be used to predict the extent or stage of stabilization.

There is a dearth of information concerning the microbiology of solid waste as stabilization occurs in a land disposal site. The organisms responsible for stabilization are ubiquitous in nature as well as in the solid waste.[6-10] Because an ever-present "seed" of organisms is available, microbial stabilization is inevitable. The rate at which it progresses, however, can vary greatly, depending on the environment. Investigators of environmental contamination currently lack specific information with regard to types and numbers of stabilizing bacteria and even of pathogenic bacteria and viruses.[11]

Examinations of solid waste and leachates derived therefrom have indicated the public health significance of solid waste. Peterson[12] isolated poliovirus type 3 and echovirus type 2 from municipal solid waste. Others[13],[14] determined that 1 gram (0.0022 lb) of residential solid waste can contain 1 million or more fecal coliforms and fecal streptococci. Soiled diapers, a source of bacteria and viruses, comprise 0.2 to 2.5 percent by wet weight of residential solid waste.[15] Other sources of fecal organisms are fomites, other sanitation materials such as household towels and rags, and animal droppings. Gaby[14] found comparable densities of fecal coliform and fecal streptococci in solid waste and sewage sludge (Figure 1). Residential solid waste immediately saturated and subjected to large daily infiltration of water yielded poliovirus in the collected leachate.[12] Another controlled laboratory study[16] indicated that the rate of moisture addition (moisture regime) is important in determining the presence of fecal indicator organisms in leachates. Low rates of moisture addition (common in sanitary landfills) tend to allow prolonged higher temperatures, which are detrimental to the survival of viruses and fecal indicator organisms.

Engelbrecht and Amirhor[17] listed three conditions that determine the potential hazard from pathogenic agents placed in a sanitary landfill: (1) the type and density of the pathogens, (2) the ability of the pathogens to remain virulent in the landfill environment, and (3) the ability of the pathogens to move through the landfill into the surrounding environment. They also found that (1) sanitary landfill leachate can significantly inactivate both bacteria and viruses, (2) pH and temperature of the leachate have a significant influence on inactivation, (3) dilution of the leachate appears to proportionately reduce its biological inactivation properties, and (4) age of the material in a fill affects the chemical composition of the leachate and thus its inactivation properties. This change in characteristics with age has a more pronounced effect on bacteria than on viruses.

A study by Blannon and Peterson[18] concluded that uncontrolled leachate may pollute the soil and groundwater and may pose hazards to the public health. Because the movement of pathogens through a landfill environment and surrounding soils is highly dependent upon water movement and soil composition, landfill site selection is a critical factor in protecting public health. In a comprehensive assessment of past studies, Engelbrecht et al.[17] suggest that the migration of leachates through soils greatly minimizes the public health threat, just as it does in septic tank leaching systems and cesspools.

Since approximately 50 percent of municipal solid waste as received at a land disposal site consists of organic material subject to biological attack, solid wastes generally provide a sufficient substrate for the growth and proliferation of microorganisms. Occasionally, this growth is inhibited by a lack of moisture or of an essential nutrient or by an unfavorable environmental stress (pH, toxicity, temperature, etc.).

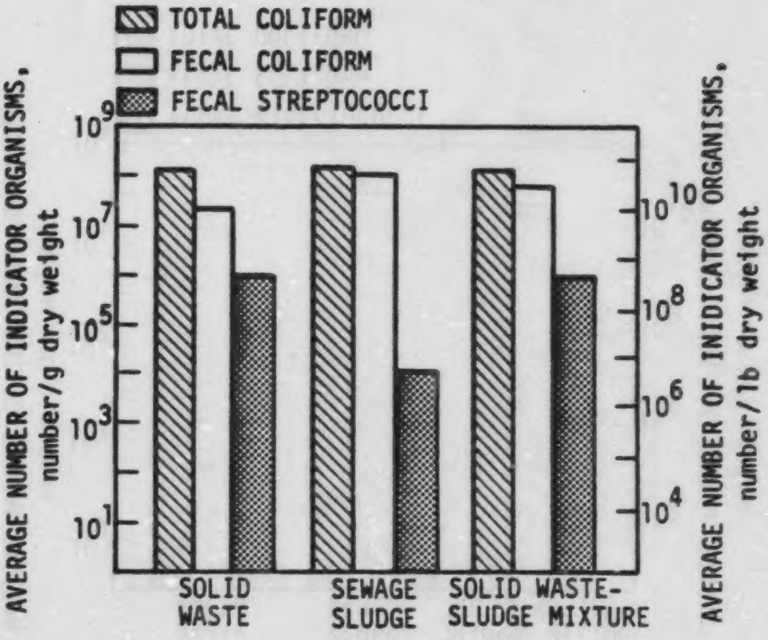

Figure 1. Indicator Orgamisms Found in Waste Streams.[14] (average of six to eight determinations)

Except for the concentrations of oxygen associated with the initial handling and placement of the solid waste in a landfill or with provision of forced ventilation at the site,[19,20] the biological decomposition of the organic fraction is accomplished principally by facultative and anaerobic microorganisms in the absence of free oxygen. This fact is paramount to an understanding of the stabilization processes that occur within a landfill and the differences in quality and quantity of the gases and leachate produced.

Unlike reactions mediated by aerobic microorganisms in the presence of free oxygen with the release of energy as heat, reactions occurring in the anaerobic environment are generally endothermic; therefore, the variations from ambient soil temperatures are slight. Temperature variations do occur in landfills, resulting from the initial aerobic biologic activity and a few intermediate exothermic reactions. The primary cause of the thermal difference between aerobic and anerobic stabilization of organic materials is that much of the potential energy of the solid waste is converted in the latter process into high-energy methane; the intermediate products percursory to methane production involve reactions that are predominantly endothermic.[21]

Observations of methane produced by landfilled solid wastes provide evidence that anaerobic conversion of complex organic materials proceeds through a series of steps essentially analogous to those that occur when sludge is biologically stabilized during anaerobic digestion. Investigations[22] of this process indicate that the sequence of events is similar--an initial liquefaction of complex organics, production of acid intermediates, and conversion of intermediates into methane and carbon dioxide. These events proceed simultaneously, are mutually dependent, and are accomplished by different species of organisms. After liquefaction has been accomplished, methane production normally is considered the rate-limiting step, since the organisms responsible for this process grow more slowly than the acid formers and are more sensitive to adverse environmental stress.

The production of acids and methane during anaerobic conversion (referred to as "acid fermentation" and "gas fermentation") is most useful in explicating the origin and fate of pollutants commonly detected in the leachate and gas that emanate from solid waste disposal sites. Excluding the associated production of biomass and assuming that microbial assimilation is favorable, one can illustrate the anaerobic conversion process by the following simple oxidation-reduction sequence:

Acid fermentation (oxidation):

$$4C_3H_7O_2NS + 8H_2O \rightarrow \underline{4CH_3COOH} + 4CO_2 + 4NH_3 + 4H_2S + 8H^+ + 8e^-$$

Methane fermentation (reduction):

$$8H^+ + 8e^- + \underline{3CH_3COOH} + CO_2 \rightarrow 4CH_4 + 3CO_2 + 2H_2O$$

Anaerobic fermentation (oxidation + reduction):

$$4C_3H_7O_2NS + 6H_2O \rightarrow 4CH_4 + 6CO_2 + 4NH_3 + CH_3COOH + 4H_2S$$

An amino acid, cysteine, was arbitrarily chosen in this reaction sequence to represent a possible but specific combination of elements commonly present in organic waste materials. The reactions illustrate the sequential anaerobic conversion of organic carbon to either carbon dioxide or methane and the associated release of reduced forms of nitrogen and sulfur. The half-reactions of the oxidation-reduction sequence are mutually dependent; organic carbon is oxidized to carbon dioxide and/or intermediate products, such as the volatile organic acids (e.g., acetic acid) during the acid fermentation half-reaction, with concomitant reduction to methane during methane fermentation. The sum of the two balanced half-reactions describes the total anaerobic conversion process as defined.

With the exception of methane, the products of conversion shown above are weak acids or bases. Upon dissolution in the water contained in or percolating through decomposing solid wastes, these acids or bases dissociate to form buffers capable of influencing and controlling the pH. Hence,

$$xC_3H_7O_2NS + yH_2O \longrightarrow \begin{array}{l} CH_4 \uparrow \\ CO_2 \rightleftarrows H_3O^+ + HCO_3^- \rightleftarrows H_3O^+ + CO_3^= \\ NH_3 \rightleftarrows NH_4^+ + OH^- \\ CH_3COOH \rightleftarrows H_3O^+ + CH_3COO^- \\ H_2S \rightleftarrows H_3O^+ + HS^- \rightleftarrows H_3O^+ + S^= \end{array}$$

Of particular interest are the CO_2-HCO_3^--$CO_3^=$ and CH_3COOH-CH_3COO^- buffer systems, which contribute to pH control at near-neutral or at low ($\cong$4-5) pH values, respectively. Low pH values generally inhibit methane fermentation and the growth and proliferation of methane-forming bacteria. An overproduction of volatile acids or their curtailed utilization for methane production depresses the pH, hampers stabilization and removal of organic and possibly inorganic pollutants from the aqueous phase, and enhances the opportunity for pollutants to be released into the environment with the leachate. Moreover, except for possible explosion hazards, the release of methane is the most desirable means of biologically removing carbon from the solid waste and it provides some interesting possibilities for energy-recovery schemes.

ANALYTICAL INTERPRETATIONS

If measurements indicate large quantities of gas and methane concentrations in excess of 50 percent, extensive methane fermentation has been established and a significant portion of the environment of the site is apparently acceptable for the growth and proliferation of the methane-formers. In contrast, if the quantity of gas is small and methane concentrations are low, methane fermentation has not been established; it is being inhibited by some type of environmental stress, or stabilization has proceeded to or near completion. Failure to detect gas production does not necessarily indicate the absence of biological activity in the site; lack of conversion of volatile acids into gas would greatly enhance the potential for high concentrations of pollutants in the leachate emanating from the site.

To illustrate what may be anticipated with respect to leachate quality, the solid waste characterized in Table 1 was landfilled in a trench, and covered with clay loam sloped at 2 percent. Collected leachate was then examined to determine its biological treatability for removal of some of the organic pollutants. Table 2 gives some of the results of these studies.

The data in Table 2 confirm the susceptibility of leachate to biological degradation as measured by conventional parameters. The COD, BOD_5, and TOC were efficiently reduced, probably as a consequence of oxidation of the acids with a resultant increase in pH and elimination of acidity. Similar changes have been observed during anaerobic biological treatment, with dramatic reductions in pollutional strength of the leachate due primarily to the conversion of volatile acids to carbon dioxide and methane.[24]

Sufficiency of nutrients is also evident in Table 2. Concentrations of nitrogen and phosphorus (needed for biological growth) were reduced after treatment because they were incorporated into the biomass. Reductions in other constituents were probably caused by changes in pH and increases in the oxidation -reduction potential, which allowed for the oxidation and/or precipitation of iron and calcium. Similar reactions with these and other inorganic species can be expected at a solid waste disposal site as the pH and/or oxidation-reduction potential changes because of variations in biological activity. Although they are not yet well adapted to evaluation of landfills, principles and techniques from the soil sciences on physical-chemical mechanisms of transport of organic and inorganic species should provide a basis for predicting possible environmental attenuation or impairment.

TABLE 2. BIOLOGICAL TREATMENT OF LANDFILL LEACHATE[23]
(ppm except as noted)

Parameter	Concentration	
	Raw	Treated[a]
COD	15,800	360
BOD_5	7,100	26
Total carbon	4,920	310
TOC	4,600	140
pH, standard units (avg.)	5.4	8.4
Alkalinity, as $CaCO_3$	3,280	670
Acidity, as $CaCO_3$	1,580	-
TKN, as N	280	16
NH_3-N, as N	10	8.5
NO_3-N, as N	19	4.3
Total phosphorus, as P	13	0.14
Ortho-phosphorus, as P	8.7	0.02
Total iron	240	<10.0
Calcium	1,200	39
Magnesium	170	140
Chloride	540	510
Conductivity, µmho/cm	5,800	2,900

[a] Aerobic biological treatment, 10 days retention time.

The character of leachate is highly dependent on the degree or stage of biological stabilization that the organic fractions of the solid waste achieve, whether this occurs within the landfill site or in a separate treatment system. Hence, a landfill that has recently received waste materials will generally tend to accumulate moisture and will progress through an acid fermentation stage involving low pH, high volatile acids content, and abundance of leached ions. In contrast, an older landfill, in which the readily available organics in the refuse have stabilized, will likely produce leachate with a pH near neutral and essentially devoid of volatile acids; the pollution potential of leachate from an older site is much lower. The range of stabilization conditions between these two extremes emphasizes the need to design and operate a landfill so as to control leachate until the potential for impairment of ground or surface water has diminished.

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SECTION 3

ANALYTICAL METHODS

Analyses for chemical, physical, and bacteriological contaminants in leachate have generally relied on Standard Methods.[1],[2] Techniques for organic characterization have been proposed by Chian and DeWalle.[3]

Methods relying on colorimetric techniques are undesirable for analysis of leachate unless extensive dilution is practical. Identification of dissolved and suspended fractions of material is subject to great variations, depending on sample handling, which affects the oxidation-reduction potential and subsequently the distribution of dissolved and suspended matter.

Leachate derived directly from sanitary landfilled municipal solid waste is generally translucent and yellowish brown under the prevalent anaerobic reducing conditions. When the same leachate sample is exposed to air, the color may range from red, due to oxidized iron, to black, due to sulfide precipitates; finely dispersed precipitates are common. Hence it is recommended that analysts avoid coloimetric techniques, resort to sample digestion for total analysis of metals, rely on instrumental analysis, and avoid differentiation of suspended/dissolved fraction of solids.

Because sampling techniques can have a significant effect on the analytical results, the purposes of sample gathering should be clearly defined and criteria should be developed to determine what special handling and precautions are required. Samples collected from a recently installed solid wast fill undergo extensive changes immediately after collection unless strict anaerobic conditions of sampling and storage are maintained. These may involve collection under an inert gas or zero ullage to prevent inadvertent aeration of the sample. Preservation by freezing also may affect the sample; precipitates have been qualitatively observed when a sample was thawed. Bacteriological assay by use of membrane filters has been subject to occasional sample overgrowth and inability to filter large volumes; this, in turn, yielded values as much as a factor of 100 lower than those obtained in assay by the MPN tube technique. Until more definitive information is assembled, it is suggested that methods recommended by the EPA be used;[4]-[7] where these are lacking, the analyst should use methods promulgated by the APWA[8] and APWA et al.[1] Reliance on instrumental methods is preferred to colorimetric or gravimetric techniques. Storage at 5°C (41°F) under zero ullage is recommended. The analyst should be especially aware of potential matrix interferences in metal analysis.

Chian and DeWalle[9] have provided an extensive compilation of analytical methods used by U.S. researchers, consulting firms, and regulatory agencies to determine physical, chemical, and biological properties of solid waste leachate. They conducted a preliminary evaluation of those methods to measure a certain parameter that was least subject to interference. Since only the accuracy of each selected method was evaluated, the precision and sensitivity must be defined in further evaluations.

Gas sampling for subsequent analysis by gas chromatography or other technique has not been standardized. A representative sample may best be obtained with gas probes by drawing the gas under a vacuum, through a collection vessel, and through a portable gas meter. The meter indicates when the composition of gas being withdrawn is constant; it is assumed that in the constant state the sample represents the landfill gas atmosphere being drawn through the sampling line. Both catalytic and thermal conductivity detectors are used with the portable meters in landfill studies. The catalytic detector is used to measure the explosive properties of the gas mixture and generally is not applicable when methane levels are high (>15 percent) or when no methane is present. The thermal conductivity detector is most applicable when large quantities of methane are present. Both detectors may be calibrated to read in percent by volume methane and are commonly provided in the same meter with a selector switch. Both detectors are recommended for detection of methane over the range commonly found at sanitary landfill sites.

Knowledge of the characteristics of organic matter in leachate is essential to the design engineer in selecting the proper treatment method. Chian and DeWalle conducted an extensive organic analysis of leachate by membrane ultrafiltration, concentration, and gel permeation separation techniques.[3] These analytic methods are elaborate and time-consuming and thus are not practical for routine use. Some inferences regarding the nature of the organic matter can be made from results of routine chemical analyses. For example, when organic matter becomes oxidized, the percentage of oxygen-containing groups within the organic compound increases. This decreases the COD, but the organic molecule is still present, though not as readily oxidized. The more readily oxidized organic matter usually is effectively treated by biological methods. The BOD is also useful to indicate the presence of readily oxidizable organic matter. It is less reliable however, because the result depends in part on the type and amount of seed and the numbers of naturally occurring aerobic bacteria, and the anaerobic nature of leachate. Basically, chemical reagents can more accurately be adjusted for highly contaminated liquids than dilutions can be performed. Chian and DeWalle found significant reductions in COD with increasing landfill age. Key et al.[10] found similar reductions over a 2-year period. Direct determination of organic acid content can also yield information regarding biological reactivity of a particular leachate.

The oxidation of nitrogenous matter can be evaluated by measuring organic nitrogen and ammonia nitrogen. As nitrogenous matter becomes more highly oxidized, it contains less organic nitrogen and more ammonia nitrogen. The data of Merz[11] substantiate such a trend over a 1-year period.

The most difficult and least accurate measurement is for suspended solids. Results are determined largely by the extent of oxidation and resulting formation of inorganic precipitates. Hence, sample gathering and storage techniques can strongly affect the results of analyses for suspended solids, or indirectly, for dissolved solids. Measurement of volatile matter by heating a sample at 550°C (1022°F) also removes some of the bound water of the inorganic fraction and thus overestimates the organic fraction.[3]

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SECTION 4

LEACHATE PRODUCTION

VOLUME

The sanitary landfill site is a part of the classical hydrologic cycle. The governing criteria for determining the volume of leachate produced are those describing the phenomena occurring at the cover material surface. A water balance can be written:

$$W_R + W_{SR} + W_{GW} + W_{IR} = I + R + E \qquad [1]$$

where W_R = input water from precipitation

W_{SR} = input water from surrounding surface runoff

W_{GW} = input water from groundwater

W_{IR} = input water from irrigation

I = Infiltration

R = Surface runoff

E = Evapotranspiration

Infiltration can be defined as:

$$I = \Delta S_s + \Delta S_R + L + W_D \qquad [2]$$

where ΔS_s = change in the amount of moisture stored in the soil

ΔS_R = change in the amount of moisture stored in the solid waste

L = leachate

W_D = water contributed by solid waste decomposition

Proper design and operation can virtually eliminate input water from surface runoff of surrounding areas, groundwater, and irrigation. Various degrees of control can be exerted over infiltration, evaporation, runoff, and the moisture storage capacity of soils and solid waste. The volume of water produced during solid waste decomposition is generally considered negligible.

Use of the water balance has been proposed by several investigators. The volume of leachate, time of initial occurrence, subsequent flow rate, and the allowable volume of irrigation water can all be determined by use of the water balance. Remson et al.[1] provide a useful computerized moisture-routing technique. Fenn and Hanley[2] apply the water balance to hypothetical landfills in Cincinnati, Orlando, and Los Angeles. Salvato et al.[3] and California investigators[4] discuss the various factors that influence runoff and infiltration and provide guidance for determining approximate values.

Surface runoff at landfill sites can be estimated by use of the Rational Runoff calculation method. Typical distributions between runoff and infiltration for a 25-mm (1 in) storm are provided in Table 3 for a variety of soils and surface conditions. The influence of slope, surface condition, and soil type on the quantity of runoff and the potential for leachate production are clearly demonstrated in Table 3. Control of slope, soil type, and surface condition (vegetation and bulk density) can reduce infiltration by as much as 54.8, 62.5, and 55.0 percent, respectively. A silt or clay loam in a pasture land area with a 5 to 10 percent slope is generally recommended to encourage runoff, limit erosion, and prevent soil shrinkage problems. A portion of the estimated infiltration may be lost by evaporation and transpiration. The remainder goes first to meeting the moisture-retention (storage) capacity of the soil and solid waste and then to leachate, in accordance with equation 2. Accuracy of determination of the runoff from storms by the Rational Runoff formula depends largely on the accuracy of the coefficient, C, chosen for the specific site. The formula does not specifically consider the previous moisture conditions of the site.

The hydraulic conductivity and other soil characteristics have been conveniently tabulated.[6] Estimates of maximum hourly infiltration provided in Table 4 are based on the assumptions that Darcy's law is applicable, the soils are saturated and uniform, and sufficient water is available to maintain unit hydraulic gradient. Comparison of Tables 3 and 4 indicates the necessity for careful determination of the hydraulic conductivity of proposed cover soils (differences as great at 10^3 in the same soil group) and interpretation of the results of the Rational Runoff formula.

TABLE 3. RUNOFF AND INFILTRATION FOR A 25-mm RAINFALL[a]

Surface[b] condition	Slope	Rational runoff coefficient[c]			Runoff[d] m^3/ha			Infiltration,[e] m^3/ha		
Pasture or meadow (cover crop)	%	Sandy loam	Clay or silt loam	Clay	Sandy loam	Clay or silt loam	Clay	Sandy loam	Clay or silt loam	Clay
Flat	0-5	0.10	0.30	0.40	25.7	77.1	102	230	180	153
Rolling	5-10	0.16	0.36	0.55	41.0	91.1	141	215	164	115
Hilly	10-30	0.22	0.42	0.60	56.3	107	155	199	149	102
Cultivated (no vegetation not compacted)										
Flat	0-5	0.30	0.50	0.60	76.7	128	153	180	128	102
Rolling	5-10	0.40	0.60	0.70	102	153	180	153	102	76.7
Hilly	10-30	0.52	0.72	0.82	133	184	210	123	77.7	46.1

[a] Reference 3.

[b] Pasture or meadow with a cover crop is assumed to be comparable to a vegetated final or intermediate cover. When cultivated with no vegetation it is assumed to be comparable to daily or unvegetated intermediated cover.

[c] Reference 5.

[d] 1 m^3/ha = 106.9 gal/acre.

[e] Not all water will be transmitted to refuse because of evaporation and transpiration losses.

TABLE 4. MAXIMUM HOURLY WATER TRANSMISSION UNDER SATURATION[6]

Soil description[a]	Hydraulic[b] conductivity, cm/sec	Hourly[c] transmitted volume, m^3/ha
Well-graded gravels or gravel-sand mixtures, little or no fines	$>10^{-2}$	$>3.6 \times 10^{3}$
Poorly graded gravels or gravel-sand mixtures, little or no fines	$>10^{-2}$	$>3.6 \times 10^{3}$
Silty gravels, gravel-sand-silt mixture	10^{-3} to 10^{-6}	3.6×10^{2} to 3.6×10^{-1}
Clayey gravels, gravel-sand-clay mixtures	10^{-6} to 10^{-8}	3.6×10^{-1} to 3.6×10^{-3}
Well-graded sands or gravelly sands little or no fines	10^{-3}	$>3.6 \times 10^{2}$
Poorly graded sands or gravelly sands, little or no fines	10^{-3}	$>3.6 \times 10^{2}$
Silty sands, sand-silt mixtures	10^{-3} to 10^{-6}	3.6×10^{2} to 3.6×10^{-1}
Clayey sands, sand-clay mixtures	10^{-6} to 10^{-8}	3.6×10^{-1} to 3.6×10^{-3}
Inorganic silts and very fine sands, rock flour, silty or clayey fine sands or clayey silts with slight plasticity	10^{-3} to 10^{-6}	3.6×10^{2} to 3.6×10^{-1}
Organic silts and organic silt-clays of low plasticity	10^{-4} to 10^{-6}	3.6×10^{1} to 3.6×10^{-1}
Inorganic silts, micaceous or diatomaceous fine sandy or silty soils, elastic silts	10^{-4} to 10^{-6}	3.6×10^{1} to 3.6×10^{-1}
Inorganic clays of high plasticity, fat clays	10^{-6} to 10^{-8}	3.6×10^{-1} to 3.6×10^{-3}
Organic clays of medium to high plasticity, organic silts	10^{-6} to 10^{-8}	3.6×10^{-1} to 3.6×10^{-3}
Inorganic clays of low to medium plasticity, gravelly clays, sandy clays, silty clays, lean clays	10^{-6} to 10^{-8}	3.6×10^{-1} to 3.6×10^{-3}

[a] According to United Soil Classification System.

[b] 1 cm/sec = 0.4 in/sec.

[c] Multiply by 107 to obtain gal/acre; based on Darcey's Law with unit hydraulic gradient.

Infiltration is dependent on the frequency, duration, and intensity of rainfall, as well as the previous moisture conditions of the soil. The Bureau of Reclamation has related rainfall intensity to infiltration and has accounted for differences resulting from vegetation, soil type, precipitation, and evaporation.[7] These differences are depicted in Figure 2; curves 1 and 2 are used for barren sandy loam and clay loam; curves 3 and 5 are used for vegetated sandy loam and clay loam.

The importance of vegetation in promoting infiltration is clearly shown in Table 3. The density and type of vegetation are also important in determining evaporation and transpiration. The consumptive use of water is determined largely by the vegetative-soil system, but ranges have been compiled (Table 5).

Moisture retention by soil is dependent on soil type and previous wetting. Soil will retain a characteristic amount of water against the pressures exerted by gravity and plant roots, which are referred to as field capacity and wilting point, respectively. These parameters are commonly expressed as a percent of volume or as a depth per unit depth of soil (Table 6).

Solid waste retains moisture in much the same way as soil. Although no data are available on the wilting point of waste materials, it is generally assumed that water is lost from the solid waste in significant amounts only through percolation. Table 7 and Figure 3 summarize data on the field capacity of solid waste. Composition of the solid waste, particle size, and density account for the range in values.

Because of its moisture retention capacity solid waste can be used as a receptor of liquids and sludges from water and wastewater treatment plants and from industrial processes. The following hypothetical example illustrates the quantities involved.

In a typical sanitary landfill, 592 kg/m^3 (1000 lb/yd^3) of municipal solid waste placed 3m (9 ft) deep, holds approximately 760 mm (30 in.) of water per unit surface area before steady-state leaching occurs. The total moisture retention capacity (from Figure 3), will approximate 400 mm/m (4.8 in/ft); 148 mm/m (1.8 in/ft) is already present as initial moisture. Therefore, a surplus storage capacity of 252 mm/m (3 in/ft) is available before steady-state leaching might be expected. This capacity is equivalent to 252 $liter/m^3$ 51 gal/yd^3). If a landfill is located where net infiltration of rain is inevitable, the surplus storage capacity will eventually be depleted and leaching will occur. Utilizing a part of the surplus storage capacity for liquid waste disposal will decrease the time required for steady-state leaching to occur. If the landfill received a net

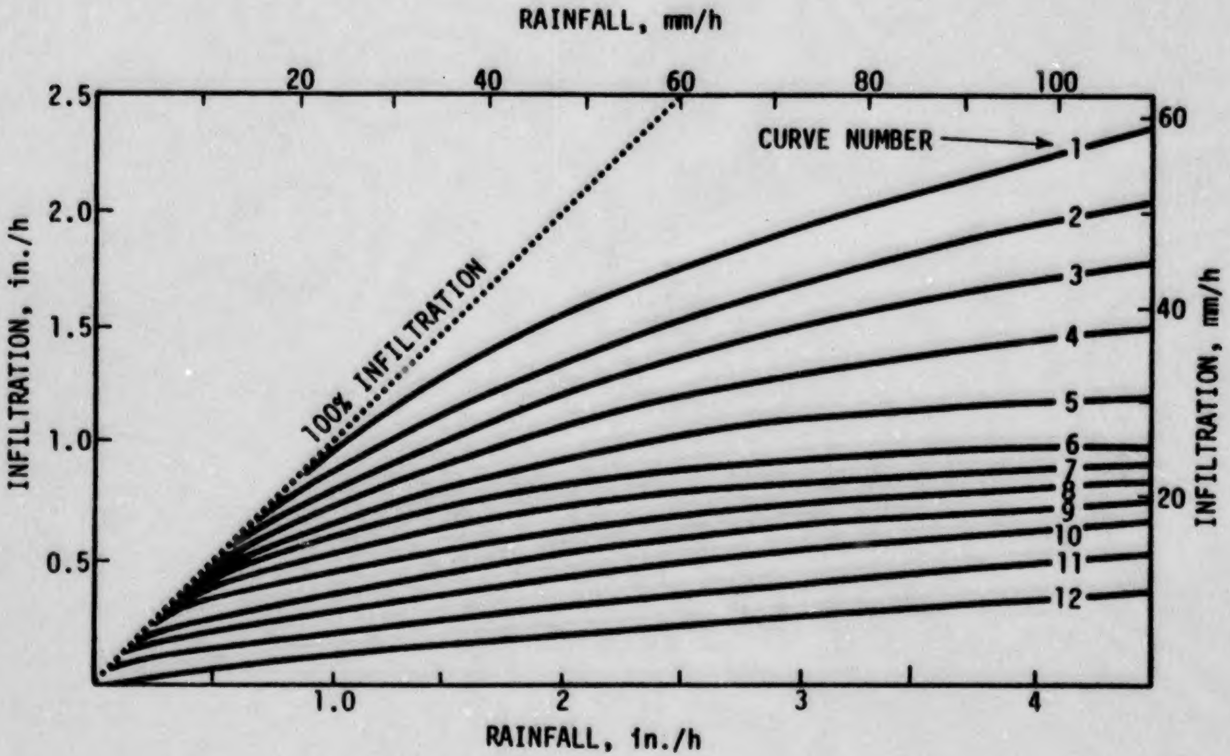

Figure 2. Relationship between Rainfall Intensity and Infiltration.[7]

TABLE 5. APPROXIMATE ANNUAL CONSUMPTION OF WATER[8] BY TYPES OF VEGETATION

	mm	(in.)
Coniferous trees	102-229	(4-9)
Deciduous trees	178-254	(7-10)
Potatoes	178-280	(7-11)
Rye	457-up	(18-up)
Wheat	508-559	(20-22)
Grapes	152-up	(6-up)
Corn	508-1905	(20-75)
Oats	711-1016	(28-40)
Meadow grass	559-1524	(22-60)
Lucern grass	660-1397	(26-55)

TABLE 6. MOISTURE CHARACTERISTICS OF SOILS

Soil	Field capacity,			Wilting point,		
	percent[a]	mm/m	(in./ft)	percent[a]	mm/m	(in./ft)
Fine sand[b]	12.0	120	(1.4)	2.0	20.0	(0.24)
Sand[c]	7.5	75	(0.9)	3.33	33.3	(0.4)
Sandy loam[b]	20.0	200	(2.4)	5.0	50.0	(0.6)
Loam[c]	25.8	258	(3.1)	13.3	133.0	(1.6)
Silty loam[b]	30.0	200	(3.6)	10.0	100.0	(1.2)
Clay loam[b]	37.5	375	(4.5)	12.5	125.00	(1.5)
Clay loam[b]	45.0	450	(5.4)	15.0	150.00	(1.8)
Clay[c]	43.3	433	(5.2)	24.2	242.0	(2.9)

[a] By volume.

[b] Reference 9.

[c] Reference 4.

TABLE 7. MOISTURE RETENTION OF MUNICIPAL SOLID WASTE

Dry density, kg/m³	(lb/yd³)	Added moisture, mm/m	(in./ft)	Total moisture retained, mm/m	(in./ft)	g/kg dry	Remarks
303.9	511.6	270.3	3.25	358.5	4.3	1179	a
313.8	528.3	245.3	2.95	360.9	4.33	1150	a
337	570	100	1.2	466	5.6	1383	b
314	530	142	1.7	438	5.3	1395	b

[a] Reference 10.

[b] Reference 11.

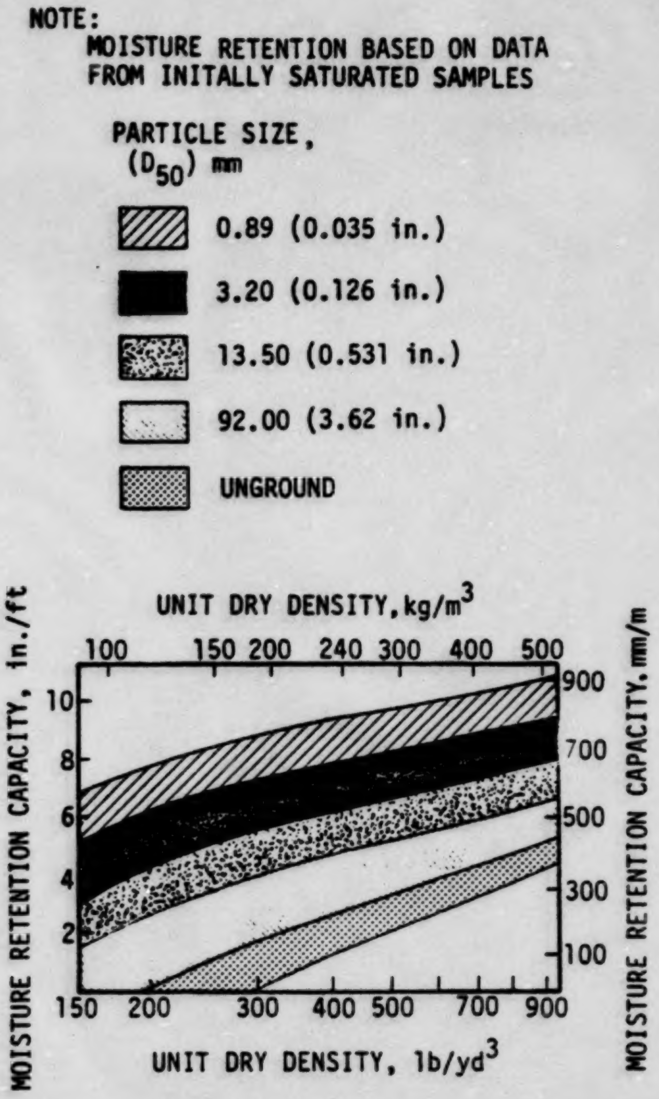

Figure 3. Moisture Retention Capacity of Solid Waste.[12]

annual infiltration of 254 mm (10.2 in) and no water were added during diposition, 3 years would elapse before steady-state leaching would occur. If 500 mm (20 in) of liquid waste was added, only 1 year would elapse. The addition of a liquid waste during deposition of the solid waste has no impact on the subsequent volumetric leaching rate.

The example shows that the production of leachate will be accelerated if water is added to the landfill, but it is not easy to evaluate how adding water will affect leachate and gas characteristics. Addition of water and the compounds dissolved in it may accelerate decomposition as well as inhibit it, may magnify or reduce the impact of leachate on the environment, and may solve or create operational problems.

The dependence of evaporation, transpiration, and infiltration on rainfall distribution and available moisture throughout the year constitutes a complex problem that has not been rigorously investigated. Remson presented a moisture routing procedure that is easily adaptable to electronic computation. Fungaroli[13] developed a national evaluation of potential infiltration based on annual averages for evapotranspiration and rainfall and no surface runoff (Figure 4). Because inclusion of surface runoff effects requires identification of site-specific characteristics, evaluation of leachate volume is not amenable to regional generalizations.[2] A California agency has provided an elaborate discussion of moisture routing, infiltration, evapotranspiration, and percolation.[14]

Hughes et al.[15] calculated from piezometer readings that 40 to 50 percent of the annual precipitation (838 mm) (34 in) in Illinois would infiltrate into landfills and produce leachate. In the dry California climate more than two-thirds of the simulated rainfall applied to a solid waste cell was evaporated.[16] The amount of runoff from large experimental landfills was measured in Sonoma County, California.[17] Figure 5 shows that with a low amount of rainfall approximately 40 percent leaves the landfill as overland runoff. At higher intensities, a constant amount of approximately 14 mm (0.55 in.) is retained at the surface while precipitation in excess of this amount appears as runoff water.

Schoenberger and Fungaroli[18] found that during the winter two landfills in Pennsylvania produced leachate at a rate of 2.9 mm/day (0.11 in/day) and 2.3 mm/day (0.09 in/day), amounts that equaled the net precipitation (rainfall-evaporation) for the period. In one European study the leachate volume equaled 44 percent of the yearly precipitation.[19] More typical values are 10 to 26 percent[20] or 10 percent[21] corresponding to 0.03 to 0.10 liters/sec (0.008 to 0.026 gal/sec) of leachate per hectare or 0.3 to 0.9 mm/day (0.012 to 0.036 in/day). Klotter and Hantge[22]

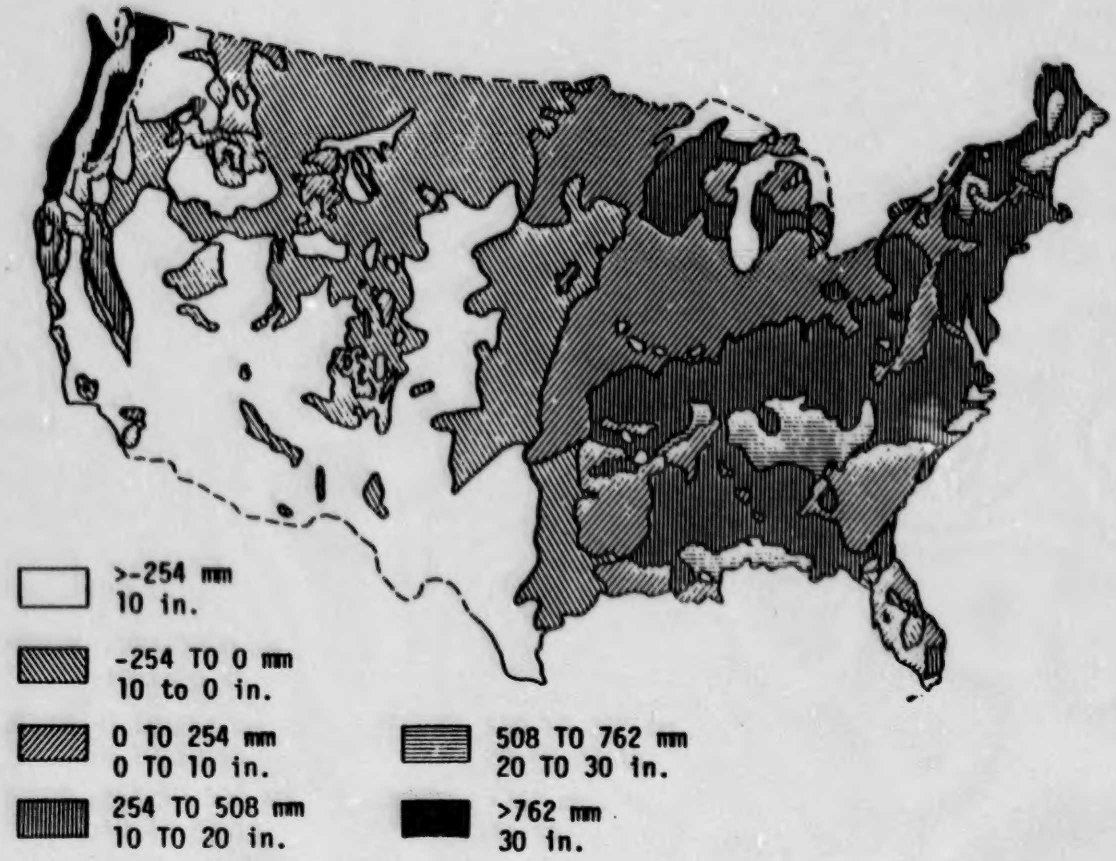
>-254 mm
10 in.
-254 TO 0 mm
10 to 0 in.
0 TO 254 mm
0 TO 10 in.
254 TO 508 mm
10 TO 20 in.
508 TO 762 mm
20 TO 30 in.
>762 mm
30 in.

Figure 4. National Patterns of Potential Infiltration.[13]

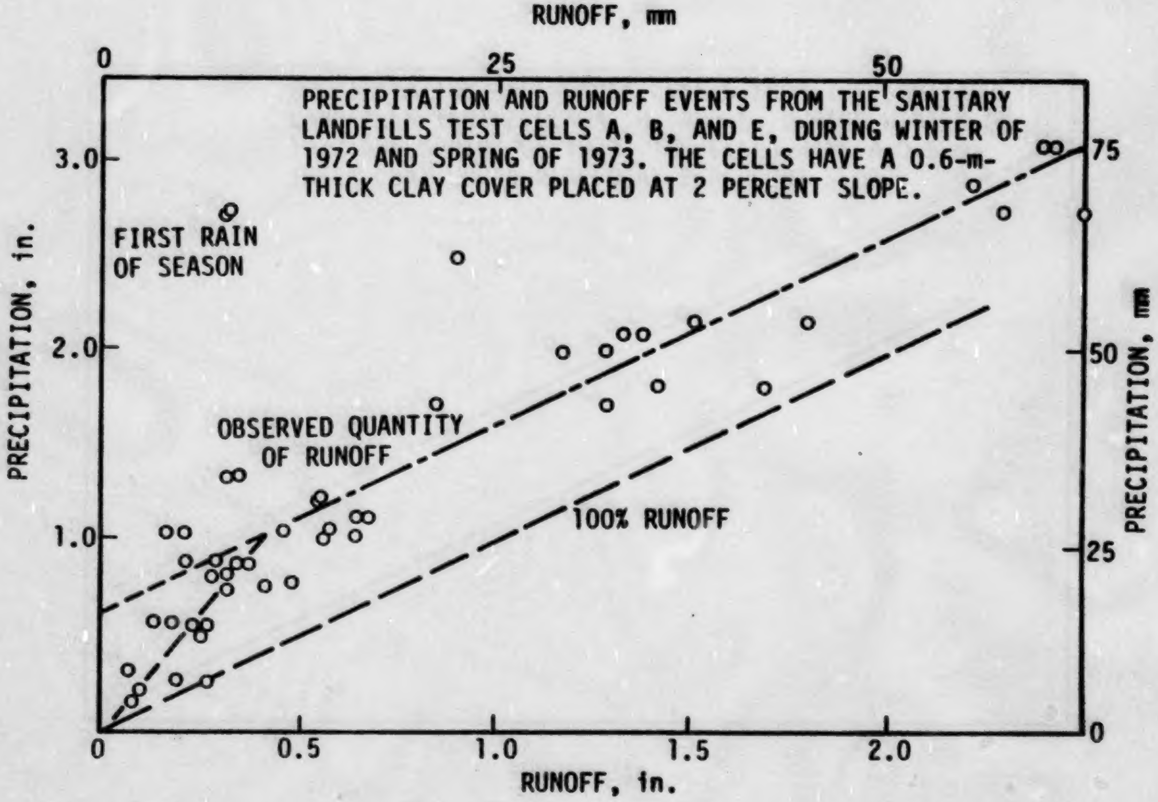

Figure 5. Runoff from Sonoma County Test Cells.[17]

measured a flow rate of 0.06 l/sec/ha (0.006 gal/sec/ac) at a 9-ha (22.5 acre) landfill; Knock and Stegman[23] calculated a volume of 0.08 l/sec/ha (0.008 gal/sec/ac). Annual rainfalls (from estimated date of reaching field capacity) of 1212.3, 1290.3, and 1377.7 mm (48, 52, and 55 in) at the Boone County Field Site[10] produced 694.2, 647.7, and 644.4 mm (28, 26, and 26 in) (estimated) of leachate or an average of 51.3 percent of the rainfall.

The reported quantities of leachate produced vary considerably. The variation may be due in part to poor identification of important factors that determine leachate production. The critical area of limited information appears to be determination of surface runoff/infiltration under the surface conditions prevalent at sanitary landfill sites. Further study of such factors as slope, erosion, vegetation, soil density, and soil type is needed.

The reported values do indicate that proper manipulation of cover material may lead to more predictable control of leachate volume. Many factors influence the rate of infiltration of rainfall into a particular soil cover. During a rainstorm many changes occur in the soil including surface compaction, swelling of clay and colloids, reduction of pore size and washing of fines into the soil pores. Storm patterns, which govern the frequency and intensity of rainfall, also affect both rate of infiltration and total infiltration.

The physical condition of the soil surface is critical to infiltration, in particular such factors as surface soil crust and vegetation cover. While surface crusting can reduce infiltration, mulches and natural covers can act to maintain a high infiltration rate. Regardless of the extent of control provided in designing and constructing cover material, a practical minimum infiltration may be expected in areas where rainfall is appreciable. The ability to predict and control the volume of leachate produced or to prevent its production in low rainfall areas is important in selecting leachate control methods, in predicting the rate of solid waste stabilization, and in determining environmental impacts.

The relative environmental significance of leachate is difficult to determine on a national basis because of the specificity of site conditions that control the moisture regime and hence the production of leachate and gas. The following hypothetical regional example is offered only for illustration.

The landfill space occupied by the solid waste produced by 10,000 people annually is estimated to be 1.15 ha-m (9.35 acre-ft); this estimate is based on a per capita generation rate of 1.5 kg (3.3 lb) (wet)/day, a landfilled solid waste density of 594 kg/m^3 (1010 lb/yd^3), and a ratio of 1 part by volume of soil cover to four parts of solid waste. Assuming the average depth of the landfill is 2.43 m (8 ft), the surface area leached each year increases by 0.473 ha (1.2 ac). If the landfill is subject to an annual infiltration of 300 mm (12 in), reported by Pierau[21] as being representative of a large part of the area east of the Mississippi (Figure 4), the leachate produced is 142 liters (37 gal) per capita/yr. The per capita leachate volume generation may be reduced by constructing the landfill to a greater depth (resulting in smaller surface area).

The quantities of materials leached from solid waste have not been extensively documented. Rovers and Farquhar[11] have summarized a few studies relating the rate of leachate collected to the depth of solid waste and reporting the mass of selected materials leached. Applying their emission rates to the situation closest to the hypothetical example indicates that at least 50×10^3 kg (110×10^3 lb) COD, 1.23×10^3 kg (2.7×10^3 lb) Cl, 1.03×10^3 kg (2.3×10^3 lb) NH_3-N, and 0.615×10^3 kg (1.4×10^3 lb) organic nitrogen will be leached from the solid waste generated annually by a population of 10,000 persons; these values correspond to 13.6 (0.03), 0.336 (0.0007), 0.28 (0.0006), and 0.168 (0.0004) grams (pounds) per capita/day respectively. Greater volumetric leaching rates, such as at the Boone County Field Site, yield greater quantities of leached materials: 42.6 g (0.09 lb) COD, 25.3 g (0.06 lb) total solids, 7.8 g (0.02 lb) EDTA hardness, and 23.0 g (0.05 lb) BOD_5/cap/day. The difference in the quantity of materials leached is due in part to the fact that data from the Boone County Field Site were obtained over a 933-day period whereas the Rovers and Farquhar data were based on only 205 days of leaching. Because the solid wastes in both of these studies are not considered to be stabilized, more materials are expected to be leached. As a reference point, the average per capita loading of COD to domestic wastewater has been reported to be approximately 57 grams (0.13 pounds).

It should be noted that the quantities cited above for materials leached from solid waste are before the leachate reacts with the soil or before any physical, chemical, or biological treatment. As the leachate migrates through the soil, these quantities are reduced, as described in Section 5. The potential quantities of material that can be leached from a landfill are of the same order of magnitude as discharge of raw sewage into sewerage systems, but in landfills the discharge is extended over several hundred days.

Fungaroli and Steiner[12] found that the quantity of materials leached per unit surface area is related to the leaching volume per unit surface area. Typical results are shown in Figure 6. The findings of this 4-year study indicate that sanitary landfills have a long-term effect on the environment.

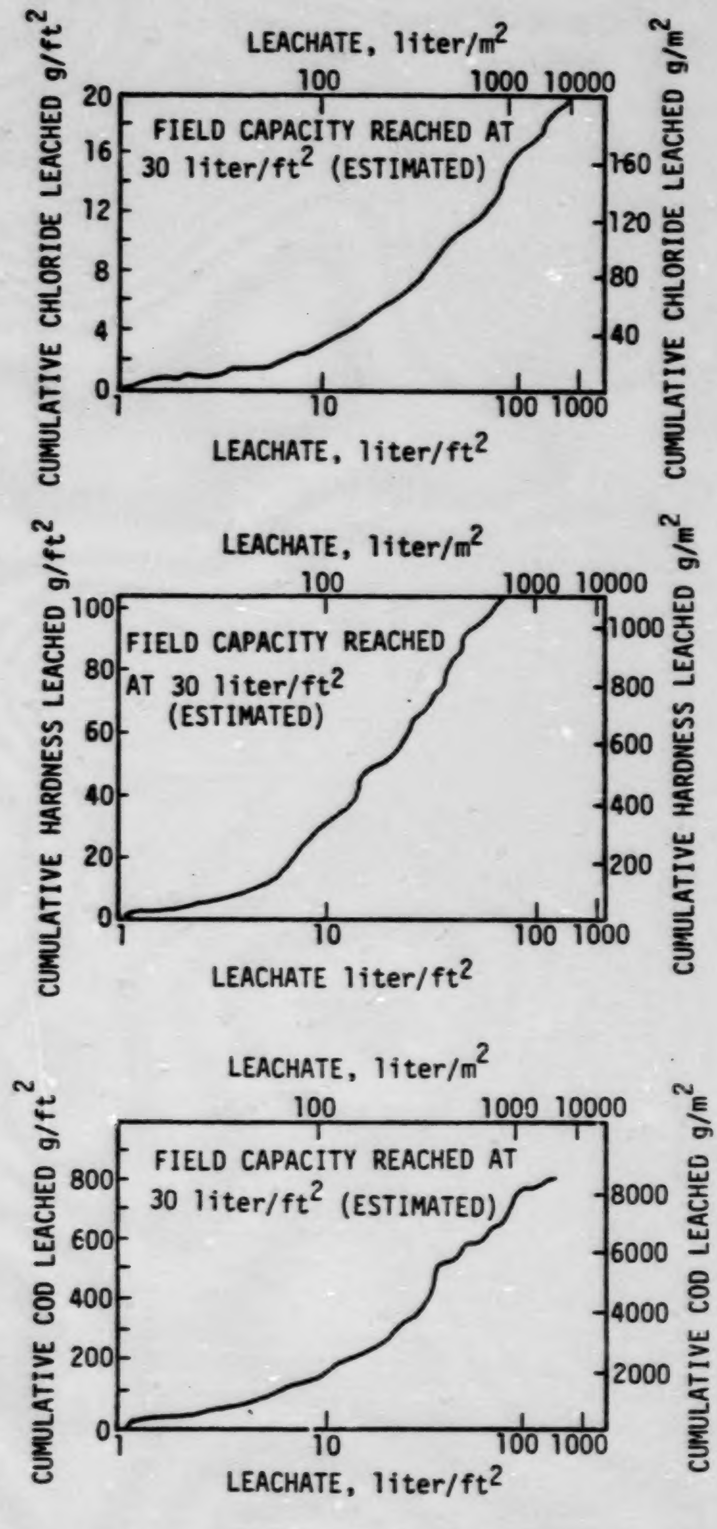

Figure 6. Relationships between the Quantity and Material[12] Leached and the Leaching Volume per Unit Surface Area.

CHARACTERISTICS

The composition of leachates reported in the literature is diverse. It also varies in individual studies over long periods of time.[10] Among the many factors that contribute to the diversity of reported compositions are time since deposition of the solid waste; moisture regime (total volume, distribution, intensity, and duration); solid waste characteristics; temperature; and sampling and analytical methods. Other contributing factors are landfill geometry and interaction of leachate with its environment prior to sample collection. Because many of these factors are rarely defined in the literature, interpretation and comparison among studies are difficult. Detailed analysis of reported values of leachate composition is given in Appendix A.

The heavy metal concentrations in leachate are generally less than 1 mg/l and the leachate pH is usually toward the acidic side of the scale (5.0 to 5.5); this significantly affects microbial and chemical reactions with the heavy metals. Ionic and nonionic materials also influence the overall solubility of a particular metal. In general, heavy metals tend to be more soluble in acid than in basic solution.

Extensive information on the bacteriologic content of leachate is not available. Fecal coliform and fecal streptococci data from the Boone County Field Site are presented in Figures 7 and 8. Engelbrecht[24] reported a similar trend of high numbers of fecal coliform and fecal streptococci initially, followed by a gradual decline to very low numbers of fecal coliform. Peterson[25] has reported isolation of poliovirus from a laboratory landfill. Thus the public health hazard of leachate discharged to a stream could be significant, since an operational landfill continually receives new pathogen-containing solid waste. The potential public health hazard from enteric infectious agents present in sanitary landfills is dependent upon three conditions: (1) number of pathogens present, (2) ability of the pathogens to remain viable in a landfill environment, and (3) migration of the pathogen from the landfill to surrounding soils and waters. The leachability of bacteria and viruses may be significantly affected by the moisture regime, which secondarily affects the oxygen content and temperature of the solid waste. Research is needed to clarify the frequency of occurrence of pathogenic organisms in leachate.

EFFECTS OF CODISPOSAL OF REFUSE WITH WASTEWATER SLUDGE

Municipal refuse may be disposed of in combination with wastewater sludge in sanitary landfills. Opinion is divided as to the need for digestion and dewatering of the sludge before landfilling. Because of operational difficulties and volume

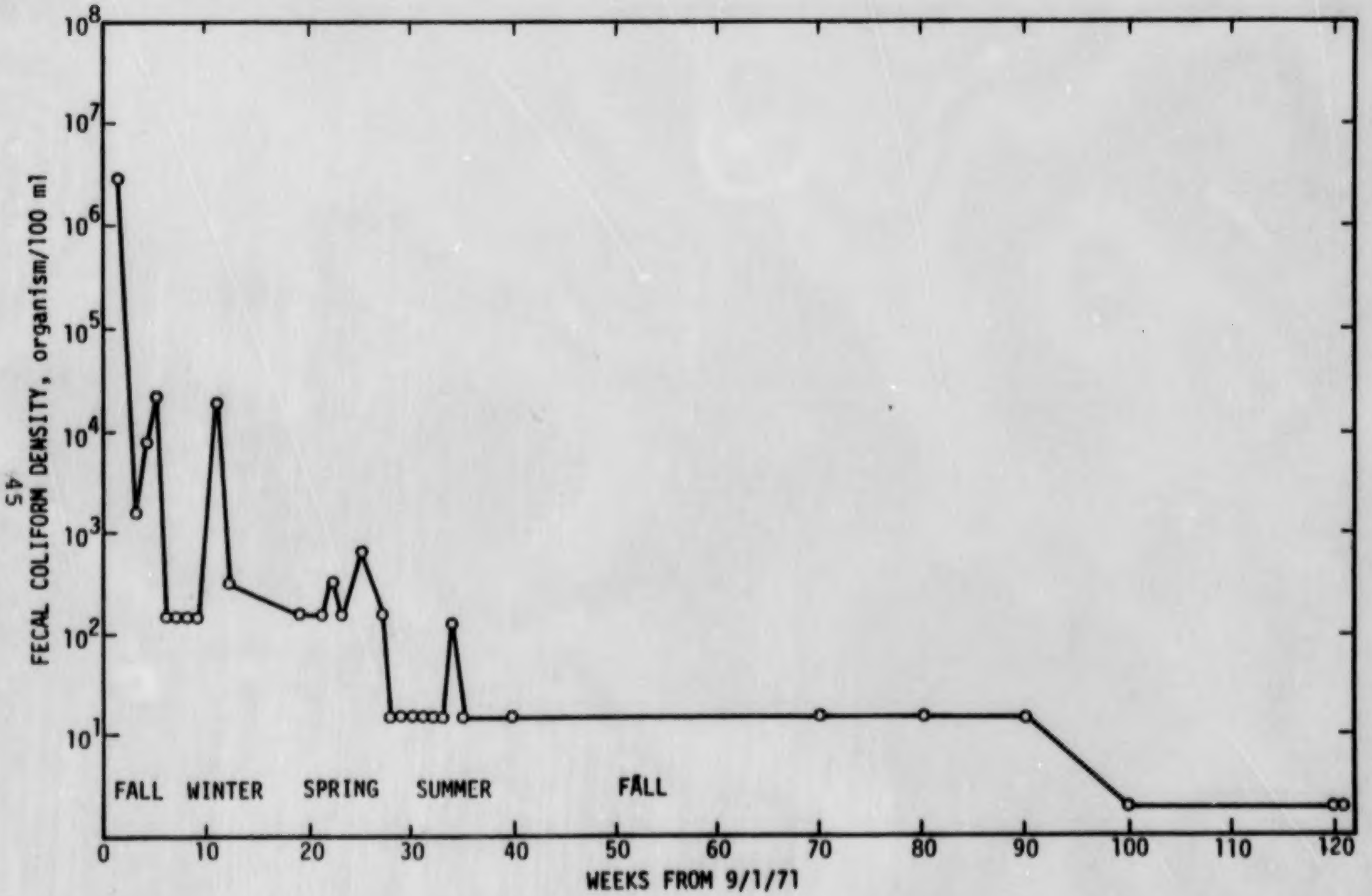

Figure 7. Fecal Coliform isolated from upper pipe of the Boone County Field Site, test cell 1.

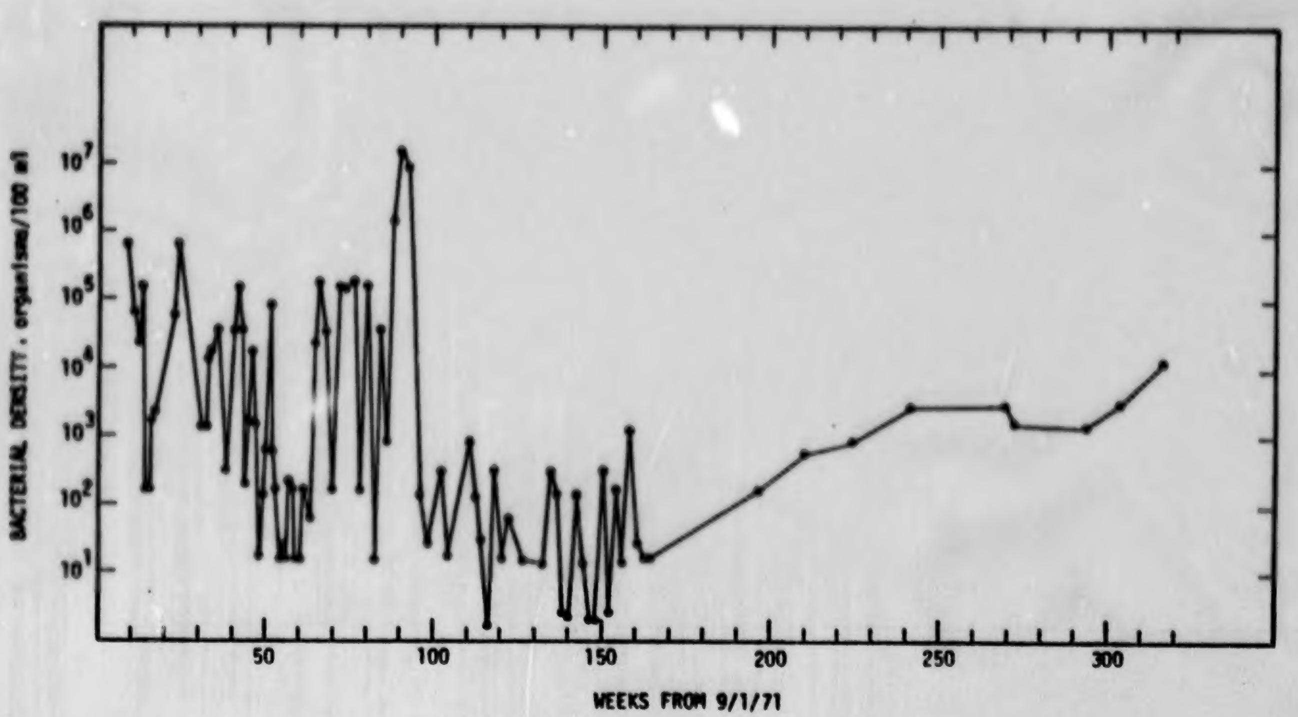

Figure 8. Fecal streptococci in leachate from the upper pipe of the Boone County Field Site, Test Cell 1.

requirements associated with liquid sludges, general practice and EPA guidelines have indicated that sewage sludge should be dewatered before landfilling.[26] Introduction of sludge into a sanitary landfill can be expected to contribute significantly to subsidence and to methane gas production.

Table 8 presents a relative ranking of potential gas and leachate production of residual wastes from various municipal wastewater treatment plants. The ash from incineration or wet oxidation of sludge has the least potential for gas and leachate production, whereas sludges from biological treatment processes have a relatively high potential.

By seeding the fill with organisms that promote decomposition, codisposal accelerates stabilization of the landfill and provides more uniformity of fill behavior. Incorporation of even dewatered sludges requires special operational techniques. The major problems are sealing the fill to prevent leaching and the health aspects of dealing with large volumes of sludge. All of these aspects should be considered in evaluation of codisposal as a management option.

TABLE 8. EFFECTS OF CODISPOSAL OF REFUSE WITH SLUDGE[26] ON PRODUCTION OF LEACHATE AND GAS

Municipal wastewater residual wastes	Relative potential leachate contamination	Relative potential methane production
Biological treatment		
Primary sludges		
Undigested	High	High
Thickened	High	High
Digested	High	High
Activated sludges		
Undigested	High	High
Thickened	High	High
Digested	High	High
Sludge cake	High	High
Ash		
Wet oxidation	Low	None
Incineration	Low	None
Chemical treatment		
Alum sludge		
Raw	Moderate	Low
Dewatered	Moderate	Low
Lime sludge		
Raw	Moderate	Low
Dewatered	Moderate	Low

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SECTION 5

LEACHATE MIGRATION AND EFFECTS

LEACHATE MIGRATION

Once significant quantities of leachate move from a landfill into the surrounding soil, the direction and rate of movement of the liquid phase and the fate of the contaminants carried in solution and suspension must be determined.

The factors controlling the movement of landfill leachate differ little from those that govern the movement of water. The principles of saturated and unsaturated flow in surface soils and in aquifers are well documented.[1-3] If soils and hydrogeology are investigated in sufficient detail and it is known how much and for how long leachate is expected to migrate, one can predict with a high degree of confidence the path and rate of movement of liquids from the landfill. Computer simulation models have been developed to aid in such predictions; one such model is described by Freeze.[4] Where the hydrogeology is complex, it may be impractical to obtain sufficient data for use in elaborate models. Simpler on-site analyses can be made to determine the location of aquifers and the rate and direction of travel of groundwater.

Most problems with landfill leachate have arisen because the hydrology of a site was ignored or was investigated only superficially for example:

1. Landfills have been located in natural recharge areas where a portion of an aquifer is so close to the surface that the leachate cannot be attenuated by passing through a significant depth of fine soils before it enters the aquifer.

2. Landfills have been located in manmade recharge areas, such as gravel pits, which cut into an aquifer so that a portion of the fill is below the water table. In such situations the leachate produced comes not only from water infiltrating downward from the surface of the landfill but also from groundwater passing laterally through the solid waste.

3. Landfills have been located directly up gradient from and close to domestic or industrial water wells, such wells have been installed close to and down gradient from an operational landfill, so that they intercept its leachate.

4. When operational landfills are monitored to determine their effects on groundwater (this is done infrequently), some monitoring wells have been located up gradient from the fill so that groundwater moving toward the landfill is sampled rather than water coming from it. Wells have been installed to monitor an aquifer deeper than the one into which the landfill is discharging.

The fate of contaminants carried in solution and suspension is affected by physical processes, such as filtration, dispersion, and mixing and by soil chemical processes, such as ion exchange, adsorption, and oxidation and reduction.

Mechanical filtration by soil is an effective process. Most suspended matter and many microorganisms are removed from leachate after a few meters of travel even through fairly coarse, sandy soils. Soils with smaller particles, through which the water flows more slowly, are even more efficient.[5-8] Because of their small size, viruses are not removed by mechanical filtration as readily as bacteria in even the finest textured soils but should be adsorbed on soil clay minerals. The extent of the adsorption and its relation to inorganic ion exchange phenomena in soils are not known. Rumero[7] and Filmer and Corey[9] indicate that there may be competition between viruses and other materials for exchange sites in soils.

Dispersion (both molecular and hydrodynamic) of leachate into uncontaminated water can reduce the concentration of pollutants in solution. This process cannot be relied on, however, because its effectiveness depends on the uniformity of the materials that make up the aquifer. Dispersion is negligible in materials that have a narrow range of grain size, but it increases as the material becomes less uniform. Lateral and vertical dispersion are about 20 percent longitudinal dispersion and even over long distances such movement will not significantly reduce concentrations in the center of a body of pollutants when the original source is wide, as landfills are.

Fungaroli and Steiner evaluated the effect on migration of three mechanisms of mass transport. They found that convective dispersion has the greatest effect in carrying leachate pollutants from the site and molecular diffusion has the least effect on migration in the direction of groundwater flow. Adsorptive chemical reactions retarded migration and were most significant in media, such as clay, containing active materials. The groundwater velocity vector was the major controlling parameter of both convective dispersion and molecular diffusion.

The flow of water in aquifers is predominantly laminar, and mixing is not theoretically significant. Mixing can be highly significant, however, when water is pumped from aquifers. Because vertical mixing in the aquifer is not appreciable, the depth of the well, the thickness of the aquifer, and the thickness of the polluted zone all affect the extent of mixing when water is withdrawn. Although flow is laminar, a leachate plume can occur at elevations other than the surface of the aquifer.

The principles governing the chemical and microbiological processes (precipitation, ion exchange, adsorption, oxidation, and reduction) that control the concentration of materials in solution in soils are well understood; these are discussed by Bear,[10] Marchall,[11] and Fuller.[12] Application of these principles to the design of soil systems for attenuation of pollutants has not, however, advanced sufficiently to allow quantitative estimates of the extent and rate of such attenuation. The soil system and the wastes that are incorporated for disposal are such complex combinations of components and processes that we can only provide qualitative descriptions and assign relative ratings regarding the ability of soils and soil materials to attenuate pollutants.

In the subsoils and earth materials below landfills the most important fraction that is active in attenuation is the clay minerals. They carry a net negative charge and therefore attract and hold cations that have a positive electrostatic charge. The ability to attract and hold cations, termed "cation exchange capacity" (CEC), differs in each clay mineral. In the montmorillonite group of minerals the CEC's range from 80 to 100 milliequivalents (meq) per 100 grams (0.22 lb) of clay; in the hydrous mica group, of which illite is the most important member, the CEC's range from 15 to 40 meq/100 g, and in the kaolinite group, from 3 to 15 meq/100 g. Properties of two minor clay groups, the chlorites and vermiculites, are not well defined, but as regards CEC vermiculite is similar to montmorillonite and chlorite is similar to illite.

By comparison, the CEC's of soils are low, ranging from 2 meq/100 g for sands to 40 to 50 meq/100 g for clays. As a first approximation the relative ability of different soils and soil materials to attenuate pollutants may be assessed by comparing their CEC's. Soils with high CEC's (high clay content) can attenuate pollutants more readily than soils with low CEC's.

It is emphasized that CEC is only an aid in relative comparisons and does not indicate the actual meq of a specific contaminant that can be removed from solution by a given weight of soil. The exchange sites probably already contain some ions. Even if they were entirely unoccupied, we could not confidently predict the removal of a particular pollutant for several reasons:

1. Different ions "compete" for the exchange sites according to their concentration in solution, their valence, and whether they form weak, ionized compounds in the soil.

2. Exchange reactions proceed at a finite rate, so that the residence time in the soil (the reciprocal of the flow rate) influences the extent of pollutant removal.

3. CEC is pH-dependent; therefore, any changes in soil pH induced by the leachate will alter the removal capacity.

4. The small amounts of hydrous iron oxides in soils exist as coatings on soil particles. Although these oxides are very active in attenuation of some metals, CEC measurements do not indicate the extent of this activity sufficiently.

In a laboratory column study Griffin and Shimp[13] found that chloride, sodium, and chemical oxygen demand were relatively unattenuated by passage through mixtures of sand and calcium-saturated clay, whereas potassium, ammonium, magnesium, silicon, and iron were moderately attenuated and the heavy metals, lead, cadmium, mercury, and zinc, were strongly attenuated. Concentrations of calcium, boron, and manganese increased over the original leachate concentrations. Precipitation, with resultant accumulation in the surface layers of the columns, was concluded to be the principal mechanism for attenuation of the heavy metals. The CEC of the clay minerals was concluded to be the dominant mechanism for removal of other substances from the leachate. In batch studies, the precipitation of heavy metal cations was a significant attenuation mechanism at pH values above 5.3, whereas adsorption was the principal mechanism for attenuation of the anions over the entire pH range studied. The adsorption and mobility of pollutants in clays were significantly affected by other nonhazardous solutes in the leachate. The pH of the leachate significantly affected the degree of attenuation. Because of the major influence of leachate characteristics, it is not feasible to extrapolate measurements in one leachate/soil system to predict behavior of contaminants in another system.

In addition to the CEC-related attenuation of pollutants in leachate, physical adsorption on surfaces is an important mechanism, particularly as regards bacteria, viruses, and organic molecules. Again, the clay minerals are the most important fraction of the soil. The specific surface area for the mineral groups, expressed as square meters per gram of caly, follows the same trend as their CEC's: montmorillonite 700 to 800, illites 100 to 120, kaolinites 5 to 20.

Another important attenuation process is related to the aeration status and the soil's microbiological flora. The organic and inorganic materials in landfill leachate are reduced products of anaerobic decomposition. If the soil below the fill is unsaturated and there is an opportunity for oxygen to laterally diffuse, these products can be further degraded or changed by aerobic, oxidizing processes. Organic compounds can be degraded to simpler compounds or even completely broken down into carbon dioxide and water; inorganics, such as iron, can be oxidized to form less soluble compounds. The extent to which these processes take place, if at all, depends on the existence of an unsaturated zone beneath the fill and on the rate of oxygen diffusion into the zone relative to the rate of consumption in microbial metabolism. In most soils the rate of oxygen diffusion under the landfill is presumed to be lower than the rate of oxygen consumption.

This brief discussion of the attenuation processes is meant to indicate the complexity of the system and some of the problems involved in quantitative prediction of attenuation. Boast[14] and Kurtz and Melsted[15] review the state-of-the-art of tracing and predicting the movement of chemicals in soils; rigorous quantitative predictions are possible only when the soil-water-soluble system is well defined (e.g., with a limited number of ion pairs and a uniform soil). Although some of the models described by Boast[14] could handle a system that includes landfill leachate, the modeling would require such detailed characterization of the system that this approach is not yet feasible.

The alternative is to make qualitative predictions of the relative ability of various soils to attenuate materials passing through them in solution. Considering the importance of the clay minerals, we can safely say that soils containing significant amounts of any type of clay (fine textured soils) are more effective attenuators than those with very little clay. Moreover, in soils containing similar amounts of clay, the attenuation capabil ity may differ with the types of clay minerals; montmorillonite -vermicullite are the most effective, and kaolinite the least. Fine-textured soils are preferred because they contain greater amounts of clays having a large surface area and high CEC's and because water usually moves through them more slowly, allowing more time for adsorption, precipitation, and ion exchange to take place. Qualitative predictions based on such considerations are useful in selecting the best of several alternative disposal sites; such factors do not indicate the amount of attenuation to be expected.

In general, attenuation is not permanent. Adsorption and ion exchange are reversible and part or all of the pollutants held by these mechanisms can be released if conditions change (e.g., if some strongly acidic or very salty wastes are disposed of at the site or if the amount of water passing through the soil increases sharply.) Consequently, land disposal systems, even at sites having extremely favorable soil conditions, should be thought of not as a final solution to disposal problems but as one method for the controlled release of pollutants into the environment.

Field and laboratory data indicate that attenuation does occur and suggest the extent. The largest body of data relates to disposing of wastewater treatment sludges on surface soils and indicates that such soils can be quite effective. This information is not directly applicable to the landfill leachate because surface soils are generally aerobic and contain significant amounts of organic materials having high CEC's, whereas the soils beneath landfills are usually anaerobic and contain little or no organic materials. Additionally, the chemical forms of the materials in wastewater sludges and landfill leachate differ enough that they probably are not attenuated similarly in similar soils.

Investigative work on subsoils and landfill leachate has been scarce, particularly that in which the soil system and the leachate are well characterized. The most useful are described below.

Leighton and Blanc[16] conducted a laboratory study on the attenuation of landfill leachate by soils of undetermined mineralogy ranging in texture from sand to silt loam. The potential cation removal ranged from 68 to 340 gram equivalents per cubic meter. These estimates were for maximum short-term removal; the long-term retention capacity would undoubtedly be significantly less.

Farquhar et al.[17] analyzed groundwater collected near four sanitary landfills and concluded that "fine textured soils have a high capacity for adsorption as well as a low coefficient of permeability and consequently the migration of contaminants through such soils is relatively small."

Lane[18] studied leachate migration through predominantly kaolinitic soils. The data on chloride distribution suggest that mixing and dispersion were significant, but there are strong indications that attenuation occurred because of ion exchange and/or adsorption.

Hughes, Landon, and Farvolden[19] made hydrogeologic studies of five landfill sites developed on glacial till containing clay minerals predominantly of the illite group. They gave the following estimates of leachate attenuation to be expected in glacial tills of Northeastern Illinois:

> "Silty and clayey tills, unfractured shales, and clays should reduce the total dissolved solid content of leachate by one or two orders of magnitude in traveling a distance of 1.5 meters (5 feet). Sands and silts will reduce the total dissolved solids in leachate by about one order of magnitude in traveling 152 meters (500 feet), and gravels and fractured rocks will be considerably less efficient." They caution, however, that: "these data apply specifically to the landfill sites studied in this report and generally to conditions in Northeastern Illinois. When extrapolating them to other areas one should consider, among other things, the groundwater gradients involved and the mineralogy of the materials through which the leachate moves - two factors that would affect contaminant and attenuation. These data should, therefore, be used with some discretion."

Although the reports cited above substantiate that the concentration of materials dissolved in leachate is attenuated by passage through soil, they do not support quantitative predictions (either for the systems that were studied or for other field situations) of the depth of soil below a landfill or the thickness of a clay liner required to prevent groundwater pollution. They do indicate the soil conditions in which a significant amount of the materials in the leachate probably will be removed, and they support estimates (based on knowledge of basic soil characteristics, such as clay mineralogy) of the relative attenuation properties of different soils.

It should be noted that pollutants have migrated readily in many instances. McGauhey et al.[6] reports that pollutants traveled several hundred meters from municipal waste disposal sites and several kilometers from industrial waste disposal sites. Saint et al.[20] report migration of heavy metals, phenols, and chlorides from waste chemical disposal pits located in a sandy soil. Changes in chloride concentration and pH were detected in an aquifer up to 300 m (1000 ft) down gradient from the pits, and heavy metals were detected 3.05 m (10 ft) below the pits. Several sharp peaks in concentration at depths of 0.46 and 1.83 m (1.5 and 6 ft) were associated with water movement during discharge periods and with changes in soil pH. Deutch[21] reports instances in which groundwater was contaminated by chromium because the soil was not an effective treatment system. This ineffectiveness was probably due to the coarse texture of the soil and the nature of the leachate, which contained high concentrations of chromium in the presence of high concentrations of other soluble salts.

Natural and synthetic landfill leachates were applied under anaerobic conditions at the University of Arizona to well-characterized samples of the major soil groups in the United States and to mixtures of the three important clay minerals. The data provide information on how well clay minerals and whole soils attenuate leachate constituents. The results substantiate the importance of clay content, hydrous oxide content, pH, and surface area per unit weight of the soil. Characteristics of the soils evaluated are listed in Table 9. The relative mobility of anions and cations in soils of different attenuation capacity are depicted in Figures 9 and 10.

The Arizona study[22] concludes that clay content and surface area are by far the best predictors of a soil's attenuation properties. The investigators found CEC less useful, probably because of the masking effect of non-CEC removal processes active in whole soils but not in a cleaner clay-sand system and also because of the neutral to slightly acidic pH levels of the soils used.

The Arizona investigators developed the following ranking of mobility of the eleven contaminants studied:

not attenuated - total organic carbon

most generally mobile - chromium^{6+}, mercury, nitrogen

least generally mobile - lead, copper

mobility varies with conditions - arsenic, beryllium, cadmium, selenium, vanadium, zinc.

Another major conclusion of the Arizona study is that characteristics of the leachate, primarily electrical conductivity and total organic carbon, significantly affect the migration of contaminants. Soils had a variable but significant capacity to retain sorbed contaminants against extraction and further leaching as shown in Tables 10 and 11.

Additionally, the available data suggest that landfill leachate may cause low-level, "secondary" pollution by displacing alkaline earth and heavy metal cations (originally present in the soil) from the exchange complex. This "secondary" pollution would be significant only early in the life of a landfill. Over a long period the concentration and composition of the landfill leachate would be the most important considerations.

TABLE 9. SOME CHARACTERISTICS OF THE SOILS USED IN RESEARCH[22] AT THE UNIVERSITY OF ARIZONA

Series	Order[a]	Soil paste pH	Cation exchange capacity meg/100g	Electrical conductivity of extract. µmho/cm	Column bulk density, g/cm^3	Surface area m^2/g	Free iron oxides %	Total Mn. ppm	Texture, %[b] sand	silt	clay	Major clay minerals[c]
Wagram	Utisol	4.2	2	225	1.89	8.0	0.6	50	88	8	4	Kaolinite, chlorite
Ava	Alfisol	4.5	19	157	1.45	6.15	4.0	360	10	60	31	Vermiculite, kaolinite
Kalkaska	Spodosol	4.7	10	237	1.53	8.9	1.8	80	91	4	5	Chlorite, kaolinite
Davidson	Ultisol	6.2	9	169	1.89	51.3	17.0	4100	19	20	61	Kaolinite
Moloaki	Oxisol	6.4	14	1262	1.44	67.3	23.0	7400	23	25	52	Kaolinite, gibbsite
Chalmers	Mollisol	6.6	26	288	1.60	125.6	3.1	330	7	58	35	Montmorillonite, vermiculite
Nicholson	Alfisol	6.7	37	176	1.53	120.5	5.6	950	3	47	49	Vermiculite
Fanno	Alfisol	7.0	33	392	1.48	122.1	3.7	280	35	19	46	Montmorillenite, mica
Mohave	Ardisiol	7.3	10	615	1.78	38.3	1.7	825	52	37	11	Mica, kaolinite
Mohave (calcareous)	Ardisol	7.8	12	510	1.54	127.5	2.5	770	32	28	40	Mica, montmorillonite
Anthony	Entisol	7.8	6	328	2.07	19.8	1.8	275	71	14	15	Montmorillonite, mica

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[a] U.S. Department of Agriculture Comprehensive Soil Classification System.

[b] U. S. Department of Agriculture system: Sand, 2 mm - 0.05 mm - 0.002 mm; Clay, < 0.002 mm diameter.

[c] The dominant mineral is listed first.

1 g/cm^3 = 0.04 lb/m^3, 1 m^2/g = 4909 ft^2/lb.

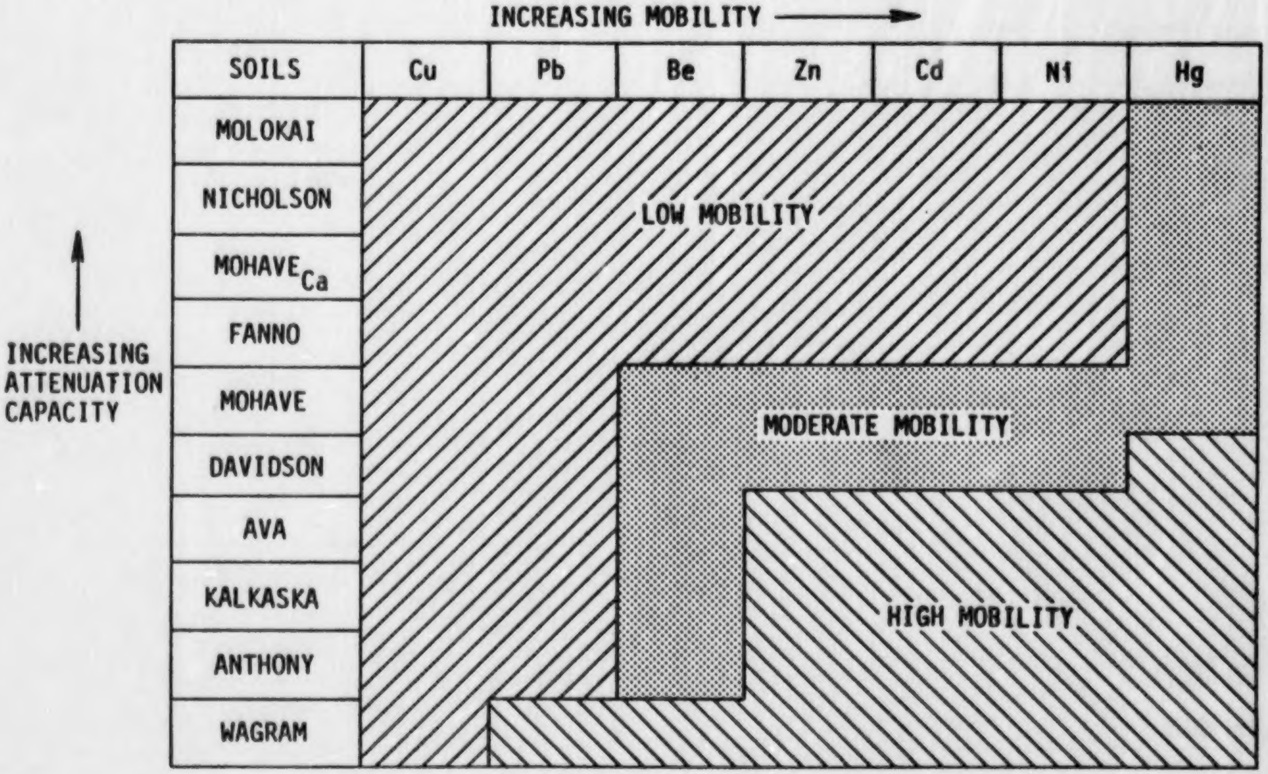
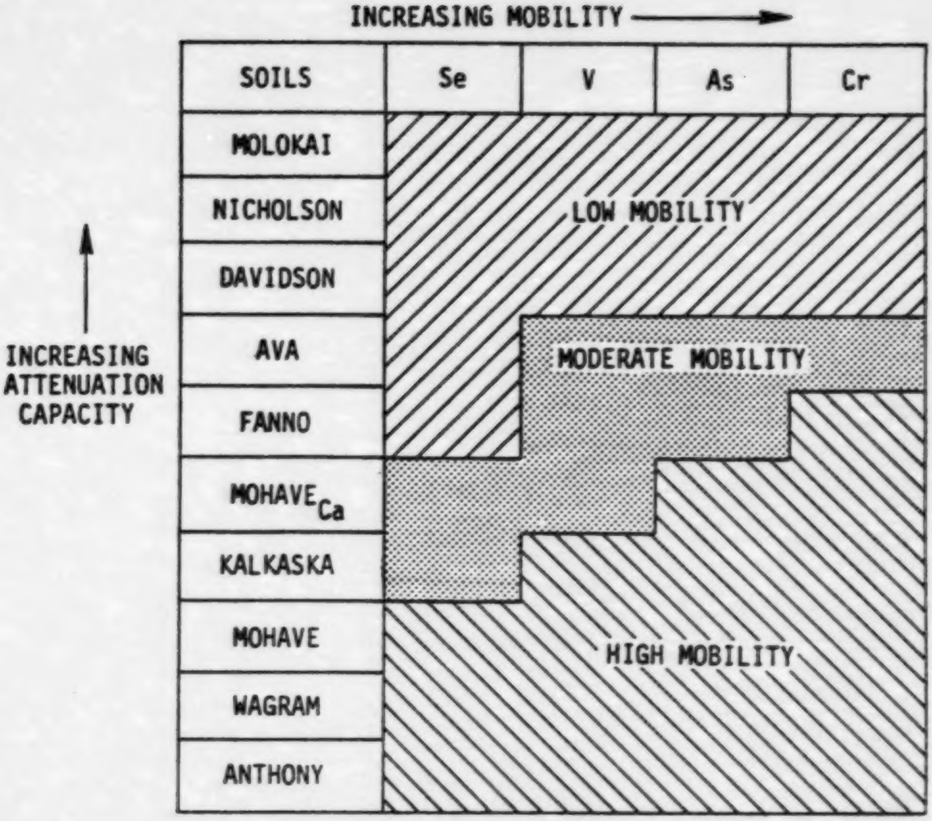

INCREASING MOBILITY
SOILS
Cu
Pb
Be
Zn
Cd
Ni
Hg
MOLOKAI
NICHOLSON
MOHAVE$_{Ca}$
FANNO
MOHAVE
DAVIDSON
AVA
KALKASKA
ANTHONY
WAGRAM
INCREASING ATTENUATION CAPACITY
LOW MOBILITY
MODERATE MOBILITY
HIGH MOBILITY

Figure 9. Relative Mobility of Cations in Soils used in the[22] University of Arizona Study.


INCREASING MOBILITY
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HIGH MOBILITY

Figure 10. Relative Mobility of Anions used in the University[22] of Arizonia Study

TABLE 10. AMOUNTS OF SORBED TRACE ELEMENTS EXTRACTED FROM DIFFERENT SOILS BY WATER (percent)

Soils	As	Cd	Cr	Cu	Hg	Ni	Pb	V	Zn
Wagram l.s.	0	9	0	0.5	1.6	7	0	9	8
Ava si.c.l.	3	5	2	0.3	0	6	0.7	0	5
Kalkaska s.	2.5	2	0.5	0.5	0	18	0.3	7.5	5
Davidson c.	0.5	1	4	1.5	2.2	1	0	0.4	3
Molokai c.	0	0.5	1	0.1	0	1.1	0	0.7	2
Chalmers si.c.l.					0.5	3		3.8	
Nicholsom si.c.	2	1	10	0	0	0.2		1.5	1
Fanno c.	4.5	1	1	3	0.5	2	0.2	8	1.2
Mohave s.l.	0	2	0	0.5	0	4	0.4	0.3	2.5
Mohave$_{Ca}$ c.l.	11.5	0.2	6	0.2	0	0.1	0.1	23	0.1
Anthony s.l.	0	3.5	0	3	9	18	0	28	2.8

Abbreviation	USDA Textural Classification
c.	clay
c.l.	clay loam
l.s.	loamy sand
s.	sand
si.c.l.	silty clay loam
si.c.	silty clay
s.l.	sandy loam
Ca	Calcareous

TABLE 11. AMOUNTS OF SORBED TRACE ELEMENTS EXTRACTED FROM DIFFERENT SOILS BY 0.1 N HCl (percent)

Soils	As	Cd	Cr	Cu	Hg	Ni	Pb	V	Zn
Wagram l.s.	11	90	8	60	14	100	41	100	100
Ava s.c.l.	11	85	7	40	16	72	78	75	64
Kalkaska s.	18	100	16	67	31	100	70	48	100
Davidson c.	5	77	7	67	49	43	70	36	42
Molokai	0	87	5	57	59	78	59	35	100
Chalmers s.c.l.					22	57		80	
Nicholson si.c.	11	75	13	27	24	63	47	55	64
Fanno c.	24	80	8	44	17	58	58	89	66
Mohave s.l.	11	89	7	55	30	80	77	93	100
Mohave$_{Ca}$ c.l.	20	74	17	51	59	31	57	77	41
Anthony s.l.	13	100	100	68	23	100	93	63	100

Abbreviation	USDA Textural Classification
c.	clay
c.l.	clay loam
l.s.	loamy sand
s.	sand
si.c.l.	silty clay loam
si.c.	silty clay
s.l.	sandy loam
Ca	Calcareous

ENVIRONMENTAL EFFECTS OF LEACHATE

Because of the diversity of leachates, the soils through which they migrate, and the receiving environments, it is difficult to generalize regarding the environmental effects of leachate. Few cases are well-documented, probably because few concerted efforts have been made to gather the appropriate information and because record-keeping at disposal sites is usually poor.

The composition of leachate and the extent of its migration are important factors in determining its effects on aquatic and soil environments. Also significant are the operation and management of the disposal site. Surface seeps (springs) of leachate that are allowed to discharge directly into surface waters will probably change them drastically unless large amounts of dilution water are available. Surface waters generally provide some mixing to minimize the adverse effects of leachate.

Two laboratory toxicity studies in which fathead minnows were placed in equally diluted fresh leachate obtained from an experimental field site yielded different results.[23] An LC 50 (lethal concentration) of 2.1 to 2.5 percent after 96 hours exposure in the first test was reported but no mortalities occurred in the second test. The chemical characteristics of the two samples were similar, and no reasonable explanation of the results has been proposed.

Over a period of 2 years in which some 600 samples were taken, concentrations of cadmium and copper ions were especially high around a municipal landfill in West Germany.[24] Walker[25] has described some of the more serious incidents of groundwater pollution that have occurred in Illinois aquifers, including several instances in which leachate from solid waste disposal sites contaminated drinking water supplies. Fecal coliform and chemical contaminants were implicated. Mead and Wilkie[26] also identified several instances in which landfills caused pollution problems.

Anderson and Dornbush[27] installed a series of wells around a landfill in a gravel pit where waste was burned and periodically covered. The water table was 2 m (6.5 ft) below the ground. Large increases of various inorganic species were noted in the groundwater immediately downstream from the fill, but water 366 m (1200 ft) downstream was almost comparable to background water quality. The chloride ion was the most suitable indicator of leaching action. In a 3 year study by Exler, groundwater downstream from a deposit was greenish-yellow, odorous, and high in salt content.[28] The area of contamination followed the flow of the groundwater in a relatively narrow but long zone, and contamination was found up to 3000 m (9840 ft) from the deposit. Seitz et al. installed boreholes around a landfill whose trenches penetrated underlying decomposed granite.[29] Groundwater rose into the solid waste during much of the year. Groundwater beneath the landfill was highly contaminated, but no serious widespread pollution of groundwater was indicated. Electrical resistivity profiles were satisfactory in following leachate migration.

During a 4-year hydrogeological and water quality study of five landfills in northeastern Illinois, Hughes et al. found that 50 percent of the yearly precipitation infiltrated the landfill surfaces and groundwater mounds formed beneath the disposal sites.[19] The fine-textured tills of Northeastern Illinois were effective in attenuating leachate pollutants, and the investigators concluded that sanitary landfill designs in this area need not include protective measures to prevent groundwater pollution. In conjunction with these studies, Cartwright and McComas performed surveys of electrical earth resistivity and soil temperature that traced mineralized water from a landfill for more than 305 m (1000 ft).[30] The soil temperature surveys detected a halo of higher temperatures around the landfills and areas of surface recharge.

Davies[31] has presented an overview of sanitary landfills and factors influencing leachate production. One case study describes soil types, groundwater flow patterns, and leachate characteristics at a landfill site in use for 5 years. Installation of a small lagoon to treat the leachate proved ineffective and created a surface water problem that otherwise would not have occurred.

Nuttall found that drainage from three refuse tips caused gross stream pollution that resulted in massive growths of the fungus, *Leptomitus lacteus*, and a low species diversity of macroinvertebrates; at least half of the invertebrate community consisted of *Oligochaeta*.[32] He also documented the distribution and relative abundance of the macroinvertebrate species and stream characteristics at various locations downstream from five refuse tips. Culham et al.[33] studied landfill leachate in Oregon. In addition to documenting adverse effects on stream chemistry, they identified *Apodyo lactea*, *Beggiotoa Thiospira*, *Paramecium candatum*, *Aspidisca lynceus*, *Chilodonella unicinata* and *Sphaerotilus ratans* in the water.

Reports of ground and surface water pollution by leachate appear with increasing frequency. A significant problem was reported in Delaware[34] involving a landfill receiving municipal and industrial wastes. The 17 hacter (43-acre) site was subjected to 228 to 303 m^3 (60,000 to 80,000 gal) of surface infiltration per day and 379 to 455 m^3 (1000,000 to 120,000 gal) of groundwater infiltration per day. A private water company and an industrial complex had to reduce the production of their well field because of migration of the resulting leachate plume. Garland[34] has reported investigations at Islip and Babylon, New York. Selenium was traced 3.04 km (1.9 mi.) from the Babylon site and a leachate plume was found 1.1 km (0.7 mi.) from the Islip site. He also cites a problem common to the Pacific Northwest: extensive slime growth as a result of leachate contamination of streams.

Three recent studies of leachate damage reported specific causes and effects of leachate pollution.[35-37] Three cases of pollution were caused by leachate from landfills that were neither designed nor operated by currently accepted sanitary landfill practices. The damage caused by the leachate pollution included loss of domestic water wells, ruin of laundry, and corrosion and staining of plumbing fixtures. The methods of constructing and operating these disposal sites actually maximized the potential for groundwater pollution.

Two of the disposal sites were unlined sand and gravel pits. At the third site, Fox Valley, waste were placed directly on the bedrock of a creviced bedrock aquifer which led to free channel flow of leachate into the groundwater. Leachate migrated at least 1500 m (5000 ft) from the Islip disposal site to a depth of 51 m (170 ft) and contaminated 0.22 m^2 (2.4 ft^2) of the surficial acquifer. Both the Peoples Avenue and the Fox Valley disposal sites are located near rivers, so that groundwater pollution was limited to 0.16 to 0.24 m^2 (1.7 to 2.6 ft^2) and 0.04 to 0.06 m^2 (0.43 to 0.64 ft^2), respectively.

Miller et al.[38] found in 11 northeastern states about 100 cases in which groundwater had been contaminated by landfills. They hypothesize that several thousand similarly designed landfills located on similar soils represent potential sources of groundwater contamination. The problems involved introduction of toxic and common contaminants into water supply wells or groundwater monitoring wells.

Assessment of the environmental effects of leachate on surface and groundwaters points to the potential for pollution to occur if adequate controls are not provided during site selection, design, operation, surveillance, and long-term maintenance. Detrimental effects due to chemical and microbial pollutants may be manifested. The drastic change in the composition of leachate as it leaves the landfill and as observed in nearby wells is a result of the ill-defined capacity of soils to renovate polluted waters. Where leachate has been a problem, faulty engineering design was determined to be a major cause.[39] Operation of a site, however, can reduce the effectiveness of any engineering design, especially because operation affects the quantities of soils used as cover material and the proximity of the deposited waste to bedrock and seepage planes. The site must, therefore, be carefully operated in accordance with the design, and appropriate modifications must be made as field conditions indicate. Surveillance, when properly implemented, will identify any movement of leachate from a disposal site. These control considerations are discussed in detail in the following section.

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SECTION 6

LEACHATE CONTROL

Methods for control of leachate may be grouped into two major categories: (1) those associated with the siting and construction of a sanitary landfill and with management of the fill in such a way as to minimize the production and harmful effects of leachate; (2) those involving the collection of generated leachate in landfill liners and subsequent treatment by a variety of methods. This section discusses in detail the current findings on leachate control methods.

SITING, CONSTRUCTION, AND MANAGEMENT

Because the prevailing hydrologic conditions strongly affect the potential for pollution of groundwater by leachate, a landfill site should be selected to achieve a favorable mix of soil textures and mineralogy, and great emphasis should be placed on determining the flow path and rate of movement of local groundwaters.

The rate of infiltration can be manipulated in the design and construction phases. Soil types having favorable infiltration rates may be incorporated into the fill or cover. Blending of coarse and fine soils to produce a wide range of grain sizes is expensive but can impede the movement of water. For example, a clay layer that is carefully constructed with regard to slope and density can be covered with a fertile soil that has the essential hydraulic and chemical characteristics to support vegetation having vigorous growth and a high rate of transpiration. Such an engineered soil system can also include provisions for control of landfill gas.

Compaction, commonly practiced at sanitary landfills, limits infiltration by changing the physical properties of the soil cover. An impermeable membrane or barrier can be used to prevent infiltration where the need is established; the costs, however, are very high.

The design engineer and landfill operator may apply a variety of other control measures, such as diversion of upland drainage, sloping of cover material, rapid attainment of final elevations, maintenance of final grades, and use of subsurface drains and ditches to control groundwater movement.

Most of these measures, however, are directed toward minimizing the production of leachate, whereas in practice several basic operational alternatives are available in landfill management. As shown in Table 12, these options are based on the rate of stabilization of the solid waste.

The first option, which includes the measures that limit infiltration of water, as discussed above, also prolongs the time required for the solid wastes to stabilize. This, in turn, lengthens the time that the site must be maintained and monitored and thus delays the erection of buildings on the site. Although achieving greater initial density through compaction or baling will probably increase the site's bearing capacity and decrease total settlement, some degree of stabilization and settlement is inevitable. Baling, which produces more uniform density, may minimize differential settlement, a development that affects the quantity of water entering the solid waste.

The second alternative is to collect and treat the leachate by use of biological and physical-chemical methods. This practice involves a more rapid but still moderate rate of stabilization. Since the composition of the leachate depends on the state of decomposition of the solid wastes, some treatment methods may be effective only at specific times during leachate generation. Collection and treatment of leachate are discussed in detail in the following subsection.

The third operational alternative is to enhance the infiltration of rainfall in the fill to obtain a moisture content that will maximize stabilization of the solid waste. Excess leachate can be recirculated through the solid waste to conserve nutrients and water. In this option, the organic content of the leachate is stablized at a rate similar to that of the organic content of the solid waste, which functions as an anaerobic trickling filter. This method does not eliminate the long-term leaching of inorganics caused primarily by solution equilibria. In climates where infiltration exceeds losses by evaporation and transpiration, the management of leachate volume is both complex and important. The practice of shredding solid wastes seems to be effective in conserving cover material, in achieving greater bulk density, and in accelerating and amplifying leachate production. Shredding therefore promotes rapid stabilization.

These options for landfill management can be clearly distinguished only in theory; in practice, optimum landfill operation is dictated by climate, by soil type, and by other physical conditions as well as by the economics of each operation and by local and regional land-use plans.

TABLE 12. OPERATIONAL ALTERNATIVES FOR A SANITARY LANDFILL

	Advantages	Means	Disadvantages
1. Minimum solid waste stabilization rate	Little or no leachate to be controlled; minimal supervision required; low initial cost	Quality controlled operation; location in dry climate; baling; leachate control, minimum; designed prevention of leachate production through ° cover grading and vegetation • diversion of surface runoff • Use of low-permeability cover	Low stabilization rate; prolonged period of low gas production; prolonged period of settlement; long-term maintenance; long-term responsibility; long-term minimal land use; lack of predictability
2. Moderate solid waste stabilization rate	Moderate quantities of leachate and gas produced over a shorter period than in 1; positive control of leachate; landfill stabilized over a shorter period; better predictability	Quality-controlled operation; leachate control required, probably including liners and treatment; shredding of waste; gas control	Leachate treatment generally required; extensive supervision required; gas control generally required
3. Rapid solid waste stabilization	Large quantities of gas produced over short period of time (recovery?); most predictable; positive control of leachate; shortest time for stabilization and land use	Promotion of solid waste degradation by shredding, leachate modification and recirculation; leachate and gas control	Large volumes of leachate (pumping, storage, liners, etc.); high operating and capital cost; extensive supervision required; gas and leachate controls required

COLLECTION

Many materials are available for use as liners to form a barrier and divert the leachate to a collection system for final disposal. Seepage of liquids in pits, lagoons, reservoirs, and canals has been successfully controlled by use of these same liner materials; however, little information is available on the effects of landfill leachate on their properties and performance.

Flexible liners are available as either reinforced or nonreinforced materials. The selection is site-specific. A reinforcing fabric (SCRIM) is laminated between layers of liner material. Nylon, dacron, polypropylene, polyester, and fiber glass are the more commonly used scrim fabrics. Advantages of reinforced liners include higher tensile and tear strength, greater dimensional stability, greater puncture resistance, potential reduction in total thickness and greater hydrostatic load capacity. Disadvantages include low yield strength, and 30 to 60 percent higher cost than unsupported liners.

The EPA is studying liner materials exposed to leachate in simulated landfills. The test materials consist of six admix materials and six flexible polymeric materials (Table 13). These have been placed at the base of a lysimeter, covered with a protective sand layer and subjected to leachate generated from shredded refuse above the sand. Additional materials and techniques of seam sealing in the leachate saturated sand layer are also under study (Table 14).

After 1 year of exposure to landfill leachate only minor changes in the physical properties were detected in the liner materials. In no case was there a significant increase in water permeability. Admix materials lost in compressive strength, but did not increase in permeability. The polymeric membranes swelled and softened but did not become permeable or lose puncture resistance or strength. These tests are being extended to 42 months so that projections of the service lives of the various liners can be made. Selected materials (polyvinyl chloride [PVC], chlorosulfonated polyethylene, chlorinated polyethylene, ethylene propylene rubber, polychloroprene, butyl rubber, elasticized polyolefin, thermoplastic polyester, polyurethane, coal tar pitch plus PVC, and emulsified asphalt on nonwoven fabric) are being evaluated by immersion in leachate and in industrial liquid and sludge wastes. The objective is to develop a protocol or testing method to determine the compatability of specific liner materials and waste leachates. A recent U.S. EPA report on liners includes a list of manufacturers, fabricators, suppliers, and installers of liners.[1]

TABLE 13. LINER MATERIALS USED IN[1] LEACHATE EXPOSURE TESTS

Polymeric liner membranes	Thickness, mm	(mils)
Polyethylene (PE)	0.25	(10)
Polyvinyl chloride (PVC)	0.50	(20)
Butyl rubber	1.58	(63)
Chlorosulfonated polyethylene, with fabric reinforcement	0.85	(34)
Ethylene propylene rubber (EPDM)	1.28	(51)
Chlorinated polyethylene (CPE)	0.8	(32)
Admix materials	mm	(inches)
Paving asphalt concrete	55.88	(2.2)
Hydraulic asphalt concrete	60.96	(2.4)
Soil cement	114.30	(4.5)
Soil asphalt	101.60	(4.0)
Bituminous seal	7.62	(0.3)
Emulsion asphalt on fabric	7.62	(0.3)

TABLE 14. LINER SPECIMENS BURIED IN PROTECTIVE SANDS[1] OF LEACHATE GENERATORS

Membrane	Supplier[a]	System of seaming
Flexseal plain	B.F. Goodrich	Heat Seal (Factory) Firestone 6079, L-1552
Flexseal backed	B.F. Goodrich	Heat Seal (Factory) Firestone 6079, L-1552
Glass fiber reinforced roofing felt	Owens Corning	None
Hypalon BR-1165-45	Burke Rubber	Cement 280
Butyl rubber	Carlisle	Cement 8800
EPDM rubber	Carlisle	Cement 8800
Neoprene rubber	Carlisle	Cement N-100
Biostat vinyliner (PVC) J14D	Goodyear	Tetrahydrofuran (THF)
Vinyliner J13D (PVC)	Goodyear	Tetrahydrofuran (THF)
Polyliner Q1E-9 671-R-02 chlorinated polyethylene	Goodyear	Fuller SC-1556 50 Toluene, 50 THF 60 Trichlorethane, 40 Toluene Fuller SC-1554
Polyliner Q1E-9671-S CPE	Goodyear	Fuller SC-1556
Hydroliner 21D 9648-S (hypalon)	Goodyear	Trichloroethylene
Polyvinyl chloride, 0.254 mm	Firestone	Solvent 6079, Cement L-1552
EPDM Rubber	Firestone	Solvent 374, Cement MAG 1265

[a] Some supplies may no longer deal in materials listed.

(continued)

TABLE 14 (continued)

Membrane	Supplier	System of seaming
Polyvinyl chloride 0.508 mm	Firestone	Solvent 6079, Cement L-1552
Polyvinyl chloride	Union Carbide	Firestone 6079, L-1552 Tetrahydrofuran
Polybutylene	Witco	Heat seal (factory)
Butyl rubber R-02 (0.356 mm)	Goodyear	Fuller SC-1556 50 toluene, 50 THE 60 trichlorethane, 40 toluene Fuller SC-155
Butyl rubber 2.29 mm	Carlisle	8800 cement w/cold seal tape, no lap sealant
EPDM rubber 1.42 mm	Carlisle	8800 cement w/cold seal tape, no lap sealant
EPDM rubber 0.635 mm	Carlisle	8800 cement w/cold seal tape, no lap sealant
Polypropylene 0.254 mm	Corth	No seam
Thermoplastic rubber, TPR 1600E	Uniroyal	5.08 x 15.2 cm piece, no seam
Thermoplastic rubber, TPR 1900E	Uniroyal	5.08 x 15.24 cm piece, no seam
Natural rubber 3.18 mm	Cal Neva	No seam, 5.08 x 17.78 cm
SBR, 70 Dur. 3.18 mm	Cal Neva	No seam, 5.08 x 17.78 cm
Polyurethane rubber	Cal Neva	No seam, 5.08 x 17.78 cm
Neoprene, 3.18 mm 25	Cal Neva	No seam, 5.08 x 17.78 cm
Foam gasket stock 6.35 mm	Cal Neva	No seam, 5.08 x 17.78 cm

Installation of the liner is as important as selection of the liner material. Liner material costs (as of October 1973) range from \$1.1 to 6.6 per square meter (\$0.10 to 0.60 per square foot), and installation costs range from \$0.11 to 1.1 per square meter (\$0.01 to 0.10 per square foot). Cost estimates for liners in the EPA project are presented in Tables 15 and 16.[1] Ultimately the user must depend upon the integrity and experience of the liner manufacturer and installation contractor and their ability to support their warranties.

Bituminous materials also are of interest for use as landfill liners. Various lining methods have been proposed, from application of bituminous seals less than 0.2 mm (0.008 in) thick to laying an asphaltic concrete base course 0.5 to 1.5 mm (0.02 to 0.06 in) thick. Use of thin membranes requires protective layers of soils on both sides to prevent punctures and tears.

Limited data are available on the suitability of clay as a liner material.[2] Test Cell 1 at the U.S. EPA Boone County Field Site has two leachate drains--one at the base of the solid waste on top of slightly acidic, silty clay (hydraulic conductivity = 1.64×10^{-5} cm/sec), and one 460 mm (18 in) below the refuse on top of a 0.76-mm-thick (3 in) chlorosulfonated polyethylene liner. Free drainage through an air trap was provided for both pipes. Leachate collected from the clay liner was 99.8 and 99.9 percent of the total leachate produced in each of 2 consecutive years.

Any decision to control leachate by using impermeable liners at the bottom of the fill requires that provision be made for disposing of or treating the collected leachate. Hydraulic pressures from both collected leachate and groundwater should be controlled to prevent failure of the liner.

LEACHATE TREATMENT METHODS

Several investigators have found that biological methods can effectively treat leachate generated in a new fill but that physical-chemical methods achieve better results when the leachate is from an old fill. An extensive study at the University of Illinois[3] related the effectiveness of each treatment method to the composition of the organic matter in the leachate. In leachate from recently deposited waste greater than 60 percent of the organic matter consists of organic acids. Because these organics are most easily removed biologically, the use of extended aeration, activated sludge, and anaerobic filter units is appropriate. After several years the organic acid fraction decreases and the amount of substances having a molecular weight larger than 50,000 increases. Since the latter are best removed

TABLE 15. ESTIMATED COST OF LANDFILL LINERS[1]

Type of liner	Installed cost,[a] $\$/m^2$	($\$/yd^2$)
Soil - bentonite (Volcaly)[b]		
4.88 kg/m^2 (9 lb/yd^2)	\$0.86	(\$0.72)
9.76 kg/m^2 (18 lb/yd^2)	1.40	(1.17)
Soil cement		
15x2 mm (6 in.) thick + sealer (2 coats - each 1.13 $liter/m^2$)	1.49	(1.25)
Soil asphalt		
15x2 mm (6 in.) thick + sealer (2 coats - each 1.13 $liter/m^2$)	1.49	(1.25)
Asphalt concrete, dense graded paving, with sealer coat		
Hot mix - 5x0.8 mm (2 in.) thick	1.43-2.03	(1.20 - 1.70)
Hot mix - 10x2 mm (4 in.) thick	2.81-3.88	(2.35 - 3.25)
Asphalt concrete - hydraulic		
Hot mix - 5x0.8 mm (2 in.) thick	1.78-2.57	(1.50 - 2.15)
Hot mix - 10x2 mm (4 in.) thick	3.58-5.02	(3.00 - 4.20)
Bituminous seal (catalytically blown asphalt)		
4.54 $liter/m^2$ (1 gal/yd^2)	1.79-2.39	(1.50 - 2.00) (with earth cover)
Fabric sealed with asphalt emulsion[c] (Polypropylene mat sprayed with asphalt emulsion)	1.51-2.24	(1.26 - 2.87)

[a] Cost as of October 1973. Cost calculated without earth cover, site preparation, or special soil base or cover except as noted.

[b] Amount needed depends on type of soil available at or near the landfill site.

[c] Depends on size of job.

TABLE 16. ESTIMATED COST OF POLYMERIC LINERS[a,b] (1973)

Materials	Thickness, mm	(mils)	Roll goods, $/m²	($/yd²)	Installed cost[c] $/m²	($/yd²)
Butyl rubber, unreinforced	0.79	(31.3)	2.69	(2.25)	3.89-4.79	(3.25-4.00)
	1.19	(46.9)	3.23	(2.70)	3.89-4.79	(3.25-4.00)
	1.59	(62.5)	3.98	(3.33)	3.89-4.79	(3.25-4.00)
Chlorinated polyethylene (CPE), unreinforced	0.51	(20)	1.89	(1.58)	2.91-3.88	(2.43-3.24)
	0.76	(30)	2.69	(2.25)	2.91-3.88	(2.43-3.24)
Chlorosulfonated polyethylene (Hypalon) with nylon scrim	0.51	(20)	1.66	(1.39)	3.44-3.66	(2.88-3.06)
	0.76	(30)	1.99	(1.66)	3.44-3.66	(2.88-3.06)
	1.14	(45)	3.28	(2.75)	3.44-3.66	(2.88-3.06)
Ethylene propylene rubber (EPDM), unreinforced	0.79	(31.3)	2.39	(2.00)	2.90-4.10	(2.43-3.42)
	1.19	(46.9)	2.89	(2.42)	2.90-4.10	(2.43-3.42)
	1.59	(62.5)	3.83	(3.20)	2.90-4.10	(2.43-3.42)
Neoprene, unreinforced	0.79	(31.3)	2.48	(2.07)	5.27-6.45	(4.41-5.40)
	1.19	(46.9)			5.27-6.45	(4.41-5.40)
	1.59	(62.5)	3.55	(2.97)	5.27-6.45	(4.41-5.40)
Polyethylene film	0.25	(10)	0.43	(0.36)	1.08-1.72	(0.90-1.44)
	0.51	(20)	0.86	(0.72)	1.08-1.72	(0.90-1.44)
Polyvinyl chloride, reinforced	0.25	(10)	.62	(0.52)	1.40-2.58	(1.17-2.16)
	0.51	(20)	1.08	(0.90)	1.40-2.58	(1.17-2.16)
	0.76	(30)	1.59	(1.33)	1.40-2.58	(1.17-2.16)

[a] Adapted from Ref. 1.

[b] All require some soil cover, cost of which can range from $0.39-1.97/m²/m ($0.10-0.50/yd²/ft.) of depth; cost not included.

[c] Membrane in test program.

by such physical-chemical methods as coagulation and by biological techniques, a combination of the two will satisfactorily remove the organic matter. The leachate from old landfills where the solid waste has decomposed is very stable and contains mainly residual refractory materials having molecular weights predominantly between 500 and 10,000. Since this residual fraction is effectively removed by use of activated carbon and reverse osmosis, these physical methods are recommended.

Aerobic Treatment of Leachate

Boyle and Ham[4] showed that activated sludge is able to reduce the COD of undiluted leachate by 80 to 90 percent, but they encountered some foaming problems. The combined treatment of leachate and municipal sewage resulted in greater reductions; however, when more than 5 percent of the influent volume consisted of leachate the settling characteristics of the waste decreased. Lower volume percentages of leachate and higher sludge ages resulted in an effluent COD of 30 mg/liter and a Sludge Volume Index (SVI) of 60.

The studies by Foree and Cook,[5] Cook and Foree,[6] Karr,[7] and Pohland and Kang[8] calculated the Monod constants for the aerobic, activated-sludge treatment of leachate (Table 17). The data indicate a relationship between the half saturation constant, k_s, and the influent concentration. Figure 11 shows that high influent concentrations resulted in high effluent concentrations, probably as a result of the higher concentration of resistant residual organic matter in the effluent. The highest effluent concentrations were observed in a study at the University of Illinois,[9] in which the influent concentration was 60,000 mg/liter. These high effluent COD values correspond to BOD values of less than 10, confirming that most of the organic matter is present as stable refractory material.

Knoch and Stegman did a study in which the leachate was obtained from recently deposited refuse and contained a large amount of readily degradable organic matter. They found that effluent of good quality was obtained with a hydraulic detention time of 10 hours in the activated sludge unit. The effluent started to deteriorate, however, at a detention time of 6 hours.[11] The best performance was observed in a two-stage system, having hydraulic detention times of 6 and 16 hours, respectively.

Removal of heavy metals is generally greater in aerobic than in anaerobic systems. Although both forms of treatment greatly reduced the iron concentration, the reduction of calcium and magnesium content was much larger in the aerobic units.[7] A similar result was obtained in the University of Illinois study, in which aerobic treatment reduced the iron concentration from 2200 to less than 1 mg/liter. The zinc concentration of 104 mg/liter was also reduced to less than 1 mg/liter. Lower removal percentages were observed for calcium (99 percent), magnesium (89 percent), sodium (9.8 percent), and potassium (9 percent). In leachate obtained from the Boone County Field Site the concentrations of iron, calcium, and magnesium were reduced by 96, 87, and 23 percent.

Schoenberger et al.[12] found that using lime, sodium, hydroxide, and sodium carbonates to adjust the pH of leachate had no major effects in removing organic matter with activated sludge systems. The same result was observed in the Kentucky study.

One study[9] showed that when nutrient addition to extended aeration units was reduced, there was an increase in refractory materials of a higher oxidation state. They also showed that treating leachate along with municipal sewage caused the amount of refractory material in the effluent to increase and that when leachate content was greater than 5 percent, the sludge settling velocity was impaired.

These laboratory studies have not led to extensive field -scale application of leachate treatment, but this capability is provided in many recent sanitary landfill designs. Planned leachate treatment plants are commonly aerated lagoons, frequently preceded by lime treatment. Only one study tested the effectiveness of activated sludge treatment on leachate from an old fill. There was no decrease in COD after an aeration period of 148 hours.[13] Such a negative result is expected because most of the organic matter in old leachate is present as biologically refractory material.

Anaerobic Treatment of Leachate

Boyle and Ham, reporting on the anaerobic treatment of leachate, found that with an influent COD concentration of 2700 to 10,659 mg/liter and a loading of 0.96 kg COD/m^3 per day (0.06 lb COD/ft^3 per day), COD removal exceeded 90 percent. The effluent COD concentration ranged from 225 to 4125 mg/liter. The authors recorded a negligible buildup of solids in the digester, an indication of a low yield factor. The leachate used was obtained from recently deposited refuse and contained large amounts of readily degraded organic matter. An aerobic polishing unit reduced COD concentration in an effluent sample from the anaerobic digester from 740 to 445 mg/liter; this suggests that a substantial portion of the residual COD was relatively nondegradable. Studies performed by Karr[7] and Foree and Reid[10] on relatively young leachate used completely mixed reactors in which the hydraulic and solids detention times were identical. The results are shown in Figure 11 and summarized in Table 17.

Although the Monod model predicts that the effluent concentration is independent of the influent concentration, Karr[7] found some dependence because the COD determination includes some of the nondegradable substances in the leachate and in the activated sludge effluent. No dependence was found with regard to the BOD and total volatile fatty acids, which indicates that these are better measures of the degradable organic matter.

Comparison of the studies by Karr and by Foree and Reid showed that higher influent COD concentrations produced higher effluent COD concentrations, a higher maximum growth rate, and a higher half-saturation constant. The work of these investigators and of Boyle and Ham, clearly showed that anaerobic digestion is an effective way to reduce influent COD concentrations when leachate obtained from recently deposited refuse and containing readily degraded organics is treated. Preliminary results from use of an anaerobic filter and an anaerobic digester showed that the former reduced COD more effectively. At an identical volumetric loading, the unfiltered effluent COD from the anaerobic filter was only 22 percent of that from the anaerobic digester.[10] Data obtained in a study at the University of Illinois indicate that an influent COD concentration of 30,000 mg/liter can be reduced by 97 percent by use of an anaerobic trickling filter (a result similar to that obtained by Foree and Reid). No substantial further COD decrease was realized when an aerobic activated sludge unit was used to polish the anaerobic effluent.

Boyle and Ham[4] noted that several cations and anions were reduced; the largest decreases (on a percentage basis) involved manganese (98), phosphorus (89), aluminum (78), iron (83), calcium (70), and zinc (31). Karr[7] generally achieved lower reductions. All metal reductions were smaller than those obtained by the aerobic systems.

Physical-Chemical Treatment of Untreated and Treated Leachate

Several investigators who tested the effectiveness of physical-chemical methods found that treatment removed large amounts of heavy metals but only small amounts of organic matter from fresh leachate. At a lime dosage of 1150 mg/liter $Ca(OH)_2$, Boyle and Ham[4] noted only a 7 percent reduction of COD in the leachate; the iron concentration, however, was reduced from 330 to 0.5 mg/liter. Coagulation at a dosage of 1000 mg/liter of aluminum sulfate reduced the COD by 5 percent but was less effective than lime in removing iron and color. A ferric

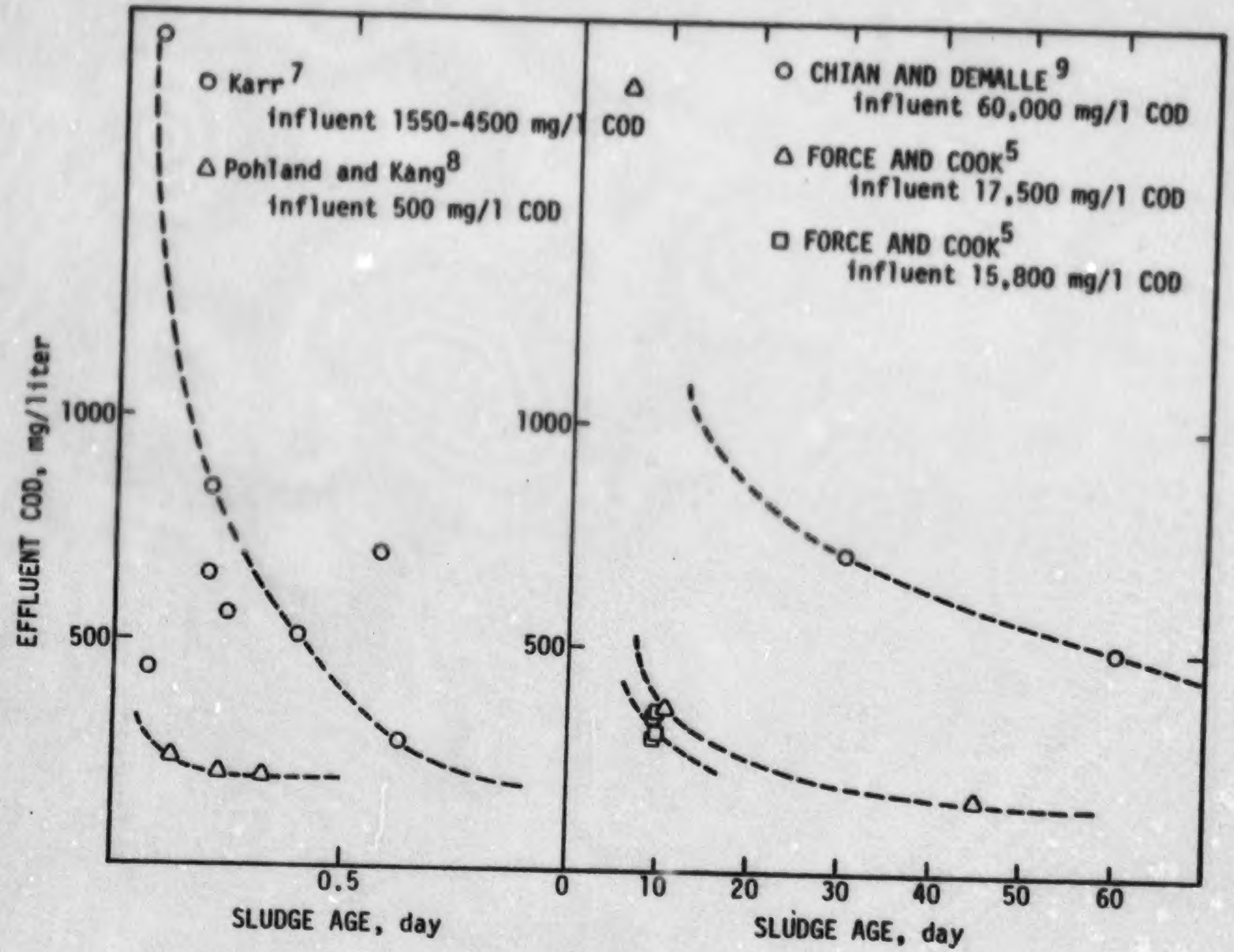

Figure 11. Effluent Cod from Activated Sludge Treatment of Leachate.

TABLE 17. KINETIC CONSTANTS OF SELECTED LEACHATE TREATMENT STUDIES

Reference	Treatment	Influent concentration mg COD/liter	Growth yield, Y mg VSS/mg COD	Decay rate, b day^{-1}	Max. growth rate, μ_{max} mg COD/mg VSS day	Substrate at $\mu = \frac{1}{2}\mu_{max}$, K_s mg COD/liter
Foree and Reid[10]	Anaerobic digester	12,900	0.33	0.17	1.39	633
Karr[7]	Anaerobic digester	6,000	0.143	0.127	0.5	300
Pohland and Kang[8] (all values as BOD)	Activated sludge unit	270	0.50	0.014	0.66	41.3
Foree and Cook[5]	Activated sludge unit	17,500	0.4	0.05	0.60	175
Karr[7]	Activated sludge unit	1550-4500	0.29[a]	0.10	1.0	1460
Chian and DeWalle[9]	Aerated lagoon	57,900	0.42	0.025		

[a] VSS determined from dehydrogenase activity.

chloride dosage of 1200 mg/liter removed 15 percent of the COD. The leachate used in the coagulation studies was from recently deposited waste and contained a large amount of readily degraded organic matter. With a leachate containing slightly more refractory organic matter, COD removal was 21 to 26 percent and iron concentrations were substantially reduced.[14] COD removals were slightly higher in leachate taken from a landfill several years old; treatment with a mixture of an aluminum sulfate and lime reduced the TOC by 40 percent. Ferrous sulfate removed only 11 percent of the organics. A lime dosage of 450 mg/liter and pH of 8.8 achieved 26 percent TOC removal in leachate from a recently shredded refuse fill.[9] In one study aeration and coagulation were used to enhance floc formation, but no substantial benefits were noted. The maximum COD removal was only 8 percent with lime treatment and even less with ferric chloride and aluminum sulfate.[15] Only one study has investigated the effects of lime on leachate from an old fill; COD concentrations did not decrease, but heavy metal concentrations decreased substantially.[13] In another study[9], lime removed organic matter having molecular weight greater than 50,000, but this fraction represents only a small portion of the organic matter leached from recently deposited waste. As biological stabilization of the waste progresses, the molecular weight of most of the organic matter leached becomes greater than 50,000 and lime treatment is very effective. As stabilization proceeds still further, the leached organic matter is composed mostly of 500 to 10,000 molecular -weight compounds. This fraction is not amenable to coagulation or precipitation by chemicals such as lime but is effectively treated by such physical methods as ion exchange, dosage with activated carbon, and reverse osmosis.

Treatment of raw leachate with activated carbon generally removes organics better than treatment with lime. Because fatty acids comprise a large fraction of the organic matter in leachate and are poorly removed in activated carbon, the overall organic removal by activated carbon is always less than that achieveable with biological systems. Boyle and Ham observed that a dosage of powdered activated carbon as great as 16,000 mg/liter resulted in a 34 percent COD removal. COD removal was 55 percent at a detention time longer than 20 minutes in a column test after two bed volumes were eluted from the column. Activated carbon enhanced the oxidation of ferrous ions, which were then effectively removed in the column. Sixty percent TOC removal was obtained in a study at the Georgia Institute of Technology using a high dosage (160 g/liter) of activated carbon.[7] Removal of volatile fatty acids such as valeric and butyric acid was greater, and removal of the more polar acids such as propionic and acetic acid was less. These results confirm other published data on activated carbon, showing that it is ineffective in treating leachate with a high fatty acid concentration.

An initial 81 percent COD removal was observed in a carbon column study on effluent from an extended aeration unit (COD ≅ 130 mg/liter) at a detention time of 15 minutes.[5] In a longer-duration carbon column study at the University of Illinois the initial TOC removal was 70 percent with a diluted leachate (TOC of 210 mg/liter) at a flow rate of 1.7liter/sec/m^2/day (2.5 gpm per ft^2/day) and a column length of 300 mm (12 in). After several bed volumes were eluted from the column, the TOC removal decreased steadily; after elution of 60 bed volumes the removal was as low as 20 percent, and after 140 bed volumes it dropped to 13 percent. Reducing the length of the column to 150 mm (6 in) decreased the initial removal from 70 to 50 percent and removal after 140 bed volumes, from 13 to 8 percent.[9]

Activated carbon reduced the COD of a well-stabilized leachate by 85 percent (from 137 to 20 mg/liter) with a removal rate of 0.17 mg COD per mg activated carbon. Results were variable, however, with different leachate samples; COD removal of another sample was only 44 percent at the highest carbon dosage. Satisfactory results have been observed at the 30-year-old Hidden Valley Landfill in Bucks County, Pennsylvania, where aeration and coagulation followed by activated carbon treatment reduces the influent BOD of 200 to 300 mg/liter to less than 10 mg/liter. Because of the high influent BOD values (compared to domestic wastewater), the carbon column had to be constructed as an upflow fluidized bed to prevent clogging by bacterial growth.[16] If a large fraction of the organic matter is refractory and the influent BOD concentration is relatively low, a packed bed downflow column is more economical.

Only two studies mentioned using an oxidant to remove organic matter in leachate. Boyle and Ham[4] found that an initial COD of 7162 mg/liter was reduced by only 6 percent at an ozone dosage rate of 12 mg O_3 per 100 ml/min. Reduction increased to 37 percent after 4 hours. Similar COD removals were observed with old leachate. COD removal was 22 percent after a 4-hour test at a cumulative ozone dosage of 1 mg O_3/mg COD.[13] Both tests, however, showed that ozonation was less effective than activated carbon in removing the organic matter from leachate.

Reverse osmosis membrane treatment of leachate generated by relatively stabilized waste produced the highest organic matter removals; TOC removal was 93 percent of 13,000 mg/liter. Most of the heavy metals were also removed. Membrane treatment, however, is effective only with well-stabilized leachates; severe membrane fouling was experienced with more biologically active leachates.[9]

The studies summarized above clearly show that physical-chemical treatment does not remove large portions of the organic matter in leachate from recently deposited waste, but that it effectively removes heavy metals, turbidity, and color. When the leachate was well stabilized and the percentage of fatty acids was small, the results were promising. Similar results could be expected from treatment of a biologically stabilized leachate in which the fatty acids have been bacterially degraded in activated sludge or anaerobic digester units.

Foree and Cook[5] found that activated carbon removed 47 to 70 percent of the COD in effluent from extended aeration units treating leachate. Chlorination removed less COD. Preliminary results by Pohland and Kang[8] showed 90 percent removal of the COD in effluent from activated sludge units by dosage with 2000 mg/liter of powdered activated carbon. Large dosages of ion exchange resins were needed to remove the ions, and the cations sodium and potassium were removed only after calcium and magnesium.

Chian and DeWalle evaluated various options for biological leachate treatment including a completely mixed anaerobic filter, a biological reactor, an aerated lagoon (extended aeration), and the combined treatment of leachate and municipal sewage in a conventional plugflow activated sludge unit.[9] They also examined the effectiveness of physical-chemical treatment steps (ozonation, activated carbon dosage, use of anion exchange resins, and reverse osmosis) in further removing organics from the biological unit effluents.

The anaerobic filter was effective in removing organic matter from high-strength leachate. The combined treatment was also effective if the leachate was less than 4 percent of the sewage flow. The rate at which the biological reactor removed organic matter depended on the substrate concentration and the operational parameters (surface area, flow, temperature). The aerated lagoon removed a high percentage of both organic matter and heavy metals. The study also showed that physical-chemical treatment is not effective in removing large quantities of organics from the leachate and that biological pretreatment is required. Several physical-chemical methods were therefore tested with aerated lagoon effluents. Ozonation removed 48 percent of the effluent TOC, and activated carbon columns removed 86 percent of the organic matter. A strong base exchange resin removed up to 85 percent of the COD. Reverse osmosis removed up to 96 percent of the salts and up to 97 percent of the organic matter.

Chian and DeWalle give cost estimates for treatment of leachates, as summarized in Table 18. The aerated lagoon provides the least expensive method of treating leachate having a comparatively low BOD_5 value (5000 mg/liter) and relatively high flow rate 1.2 liters/sec (20 gpm). At a 1.2 liters/sec (20 gpm) flow rate, as the BOD_5 level of the leachate increases treatment

TABLE 18. SUMMARY OF COST ESTIMATES FOR LEACHATE TREATMENT[17]

	Leachate gal/min[a]	Typical effluent COD, mg/liter		Costs of treatment, $/1000 gal leachate[a]	
Influent BOD, mg/liter		25,000	5,000	25,000	5,000
Activated sludge (AS), combined treatment	20	30	30	23.6	6.0
	2	30	30	41.4	11.9
Aerated lagoon (AL)	20	500	100	17.9	4.1
	2	500	100	31.6	10.0
Anaerobic filter (AF)	20	1500	300	22.1 (17.9)[b]	6.8 (5.9)
	2	1500	300	43 (38.8)	17.7 (16.8)
AL-sand filter (SF) -activated carbon (AC)	20	125	25	25.7	7.3
	2	125	25	39.9	13.7
AL-SF-AC-Reverse Osmosis (RO)[c]	20	25	5	27.6	9.2
	2	25	5	44.6	18.4
AF-SF-AC	20	375	75	32.8 (28.6)	10.6 (9.7)
	2	375	75	54.2 (50)	22.0 (21.1)
AF-SF-AC-RO[c]	20	75	15	34.7 (30.4)	12.5 (11.5)
	2	75	15	58.9 (54.3)	26.7 (25.4)

[a] 1 gal = 3.79 liter.

[b] Numbers in parenthesis indicate cost of treatment after deducting credit for methane produced at $1.50/1000 ft^3.

[c] After RO treatment, total dissolved solids decreased to 300 and 60 mg/liter with influent leachate BOD concentrations of 25,000 liter and 5,000 mg/liter, respectively.

with anaerobic filters becomes increasingly attractive, and at a BOD_5 value of 25,000 mg/liter the cost equals that of the aerated lagoon process if credit for the methane gas produced is deducted.

Well-stabilized leachate from an old landfill can be treated by similar physical-chemical methods, although preliminary results indicate a difference in their effectiveness, possibly because the nature of the organic matter in leachate varies. The studies summarized above clearly indicate the effectiveness of biological treatment of leachate when its organic matter consists mainly of free volatile fatty acids, as is the case with leachate from recently deposited waste. Leachate from more stabilized waste is better treated by physical-chemical treatment methods, because they remove more residual organics.

Biological waste treatment methods are particularly susceptible to process upsets due to shockloads, whether hydraulic or chemical. Leachate volumes are responsive to seasonal variations and possibly to major rainfall events. Because seasonal variations also affect the chemical composition of leachate, operation of an on-site biological treatment system must include some provision for flow equalization. Also, provision is required for managing the sludges produced from an on-site treatment facility.

Land Treatment of Leachate

Two studies investigated the effects of land treating leachate. Bramble[18] used 10 plots 2.14 m (7 ft) long, 0.92 m (3 ft) wide, and 0.46 m (1.5 ft) deep, with a 7 percent slope along the length and a 5 percent slope along the width. The plots, located at the EPA Boone County Field Site, were dosed twice a week at a rate of 13, 25, and 51 mm (0.5, 1.0, and 2 in.) per week of leachate with a COD of 10,000 mg/liter. COD removal percentages at these respective dose rates were 99.5; 98.5, and 85 percent. Ammonia and organic nitrogen were reduced to a somewhat lesser extent; the nitrate concentrations increased significantly--to as high as 50 mg/liter No_3-N. Removal of metals was relatively high, the iron removals ranging from 88 to 99 percent. Removals of zinc were similar, and removals of potassium, calcium, magnesium, and sodium were lower. The high rates of removal of solids and metals suggest that the permeability of the soil may decrease somewhat with time and that the leachate would then have to be lime-treated to remove these clogging substances. When land treatment is practiced on a larger scale, the frequency of leachate additions should be optimized in such a way that the nitrates formed during the aerobic period that prevails between additions are effectively removed by denitrification during the anaerobic period that occurs after leachate addition. Similar results were observed in Bluefield, West Virginia; spray irrigation, however, led to heavy coating of the vegetation by iron oxides.[19] It may be desirable, therefore, to remove a substantial part of the metals by aeration or chemical treatment before the leachate is applied. More studies are warranted, with emphasis on determining the optimum frequency of application.

Recirculation of Collected Leachate

Pilot-scale facilities at the Sonoma County Landfill[20] were used to evaluate the effects of recirculating collected leachate. This study showed that recycling is effective in removing organics from leachate, less effective in removing total dissolved solids (TDS) and Kjeldahl nitrogen, and minimally effective in removing inorganic ion. The process produced leachate with organic strength matching that of municipal wastewater, but generally with greater amounts of other constituents. Recycling appeared to be as effective as external aerobic or anaerobic ponding, with less likelihood of being adversely affected by cold weather. Leachate recycling dramatically accelerated settlement and therefore the achievement of physical stability of the fill.

In a similar study at the Georgia Institute of Technology, Pohland[21] reported promising results. Recirculation of leachate through a landfill promotes a more rapid development of an active anaerobic bacterial population of methane formers, increases the rate and predictability of biological stabilization of the readily available organic pollutants in the refuse and leachate, dramatically decreases the time required for stabilization, and reduces the potential for environmental impairment. Some of these results are depicted in Figures 12 through 14.

Leachate recirculation with pH control and initial sludge seeding may further enhance treatment efficiency so that the time required for biological stabilization of the readily available organic pollutants in the leachate can be reduced to a matter of months rather than years.

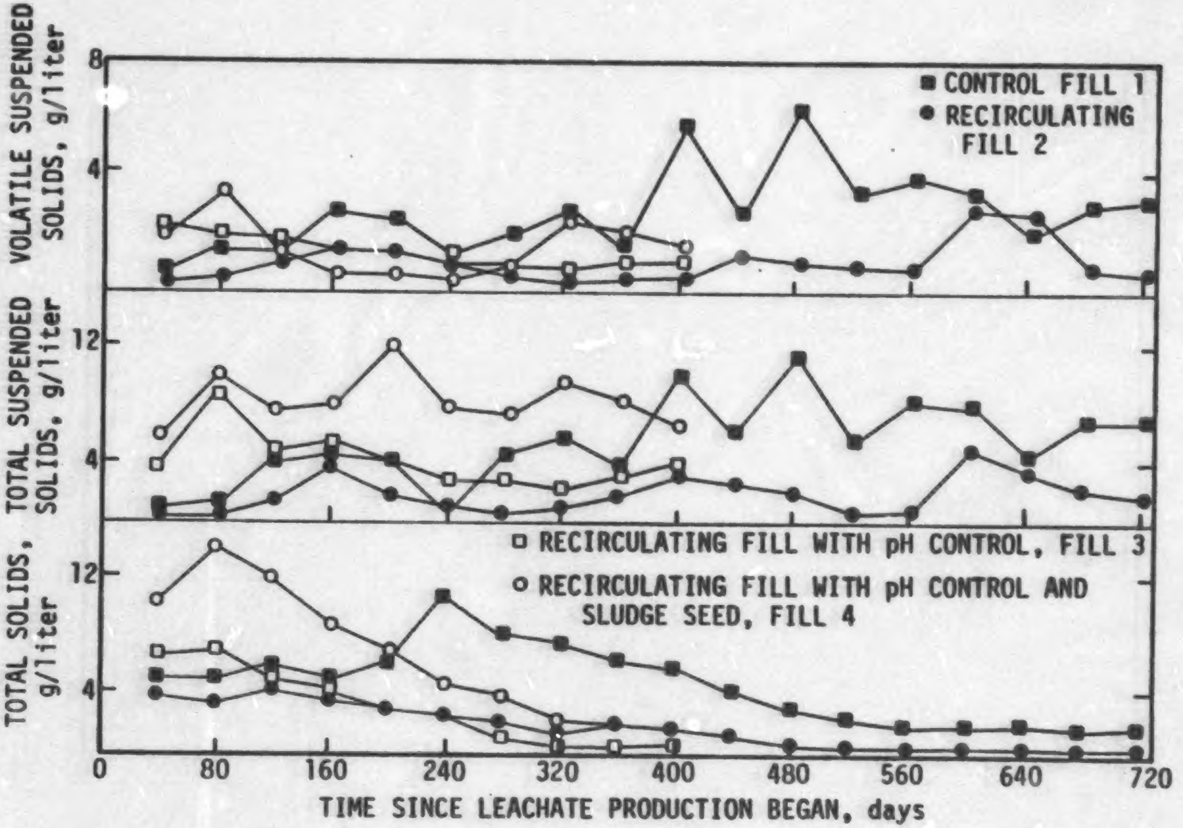

Figure 12. Solids Concentration of Recirculated Leachate.[21]

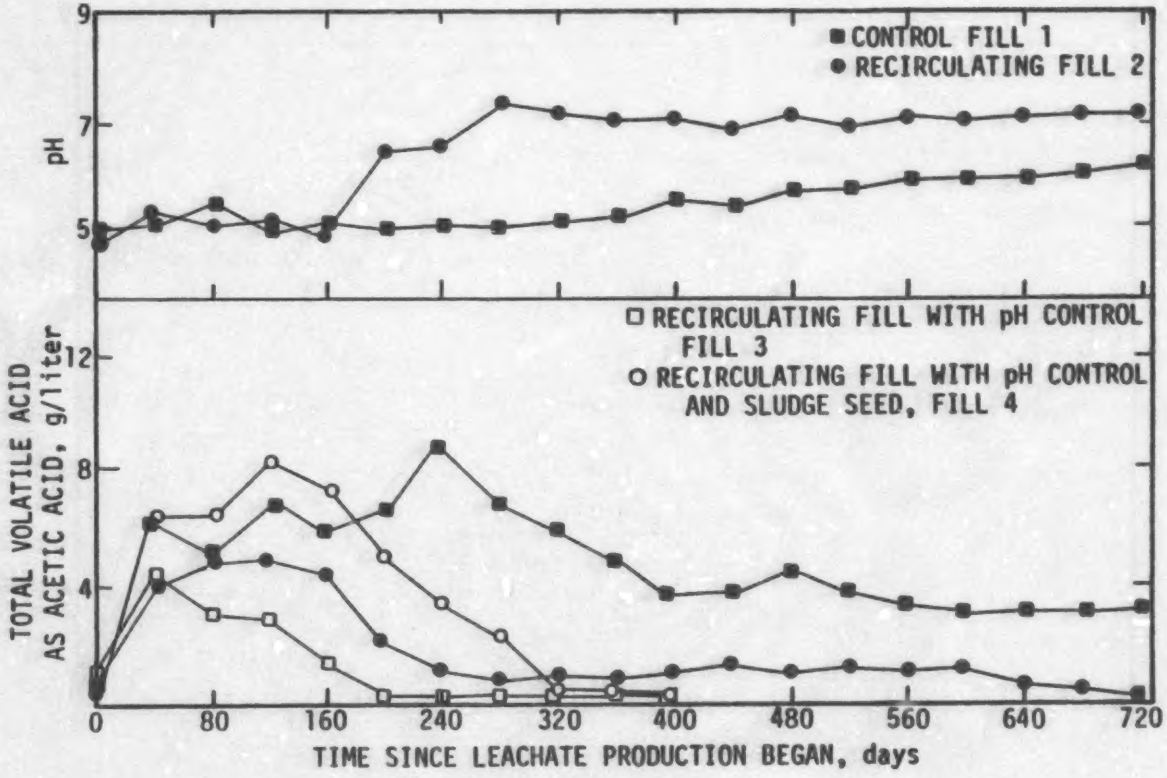

Figure 13. pH and Total Volatile Acid Content of Recirculated Leachate.[21]

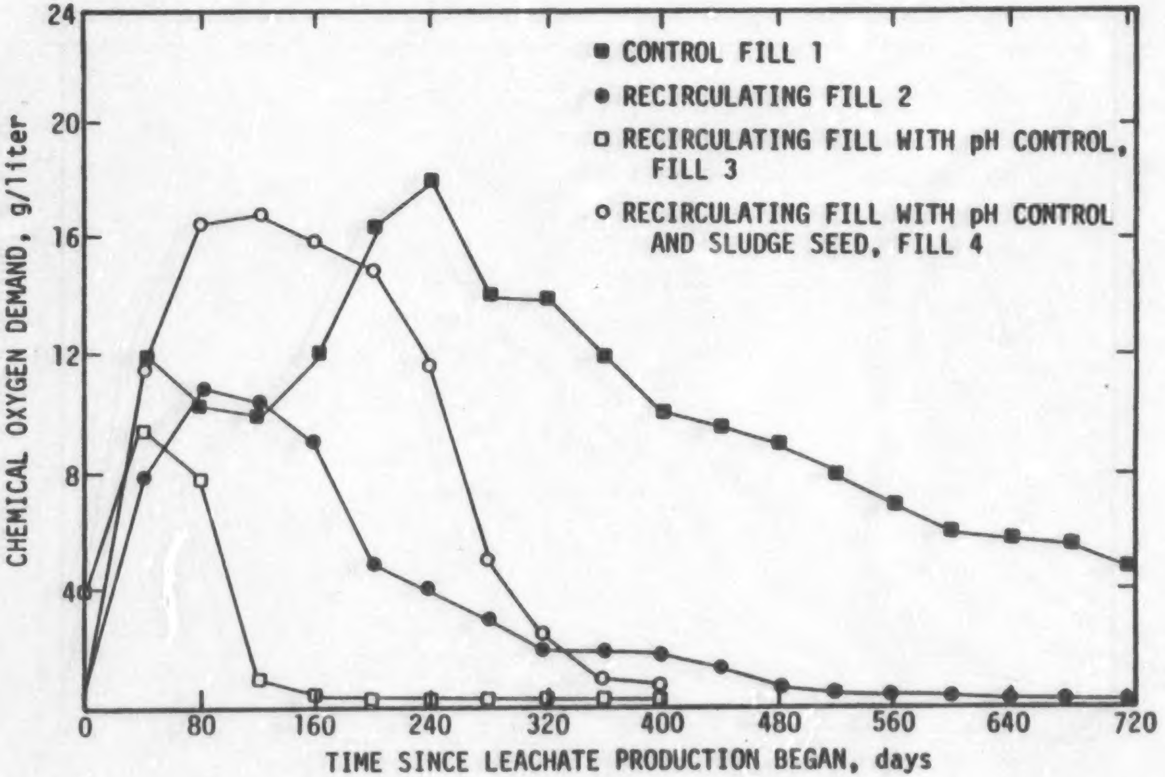

Figure 14. Chemical Oxygen Demand of Recirculated Leachate.[21]

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SECTION 7

GAS PRODUCTION

The biological stabilization of solid waste in a landfill produces gas, as described in Section 2. The environmental conditions under which biological stabilization occurs (e.g., anaerobic or aerobic environment, pH, temperature, moisture) determine the composition and rate of production of the gas. The principal gases of concern are carbon dioxide and methane, but smaller quantities of hydrogen, hydrogen sulfide, and other gaseous waste products of biological metabolism can be produced.

VOLUME

Biological decomposition of organic matter in sanitary landfills proceeds via microbial fermentations. If a complete methane fermentation occurs, the overall stoichiometry of the fermentation is constrained by the law of conservation of mass to satisfy the following reaction:

$$C_vH_WO_xN_yS_z + (v - \frac{W}{4} - \frac{x}{2} + \frac{3y}{4} + \frac{z}{2})\ H_2O$$

$$(\frac{v}{2} + \frac{W}{8} - \frac{x}{4} - \frac{3y}{8} - \frac{z}{4})\ CH_4 + (\frac{v}{2} - \frac{W}{8} + \frac{x}{4} + \frac{3y}{8} + \frac{z}{4})\ CO_2$$

$$+ (y)\ NH_3 + (z)\ H_2S.$$

If the methane fermentation does not become established, a mixture of carbon dioxide, hydrogen, ammonia, hydrogen sulfide, and small volumes of organic acids are formed. The stoichiometry of this fermentation cannot be predicted.

As a consequence of the methane stoichiometry, the decomposition of large amounts of paper results in a molar carbon dioxide/methane ratio of about 1:1. The methane fermentation of only the garbage fraction of the solid wastes yields a carbon dioxide/ methane ratio of approximately 1:2. Finally, if the methane fermentation does not become established, the gas produced should be rich in carbon dioxide and small amounts of hydrogen. The nitrogen in a gas sample is not a fermentation product but is the residue of the air that permeated the fill material during and after construction.

Anderson[1] calculated a theoretical gas production of 0.63 m^3 per kg (6.6 ft^3/lb) of dry solid waste when the waste containing 75 percent volatile matter was degraded by anaerobic bacteria. Golueke[2] reports that kraft paper, which accounted for 40 percent of the organic refuse in his investigations, is 90 percent digestible, because the lignin (a refractory organic) is removed. Approximately 33 percent of the municipal refuse he investigated consisted of newsprint, of which 50 percent is digestible because lignin is still present. Complete anaerobic digestion of the paper fraction would therefore produce 0.44 m^3/kg (7.04 ft^3/lb) of volatile solids. The remaining 27 percent of the organic refuse (food waste, lawn trimmings, and plastics) was probably 50 percent digestible. Golueke calculated that all the organic material together would produce 0.55 m^3/kg (8.8 ft^3/lb) of volatile matter. The actual volume would probably be lower because a significant amount of the carbon dioxide gas is dissolved in water. The maximum amount of gas produced from shredded solid waste by thermophilic bacteria in a completely mixed digester at a temperature of 60°C (140°F) over a 30-day detention time was equal to 0.45 m^3/kg (7.2 ft^3/lb) of volatile solids added. At 40°C (104°F), which is closer to the optimum for anaerobic mesophilic bacteria, gas production was only 0.37 m^3/kg (6 ft^3/lb) of volatile solids added.[3]

Several studies have measured the effects of moisture content on gas production and on the rate of solid waste stabilization. Popel[4] found that the moisture content of the solid waste inevitably determines the rate of temperature increase, the maximum temperature, and the amount of carbon dioxide produced when solid waste is degraded aerobically by thermophilic organisms. The temperature fluctuations in the decomposing mass were closely related to variations in oxygen availability. Straub[5] noted that when the oxygen content of solid waste decreased to very low levels, anaerobic processes replaced the aerobic processes. When production of organic acid and formation of methane and carbon dioxide began, the temperature decreased substantially, as did the rate of stabilization. Horstmann et al.[6] also found that conversion from aerobic to anaerobic conditions reduces the generation of heat and that the decomposition therefore proceeds more slowly. Teenstra[7] states that the maximum carbon dioxide content in the overlying gas is 20 percent during aerobic decomposition of refuse because the amount of carbon dioxide formed is equal to the amount of carbon and oxygen consumed. When the carbon dioxide exceeds 20 percent, anaerobic conditions prevail.

Carpenter and Setter[8] found relatively high concentrations of carbon dioxide and methane (up to 34 and 64.3 percent, respectively) at 0.6 m (2 ft) depth in a landfill where the water table was 1.8 m (6 ft) below the covered surface. They noted much lower concentrations in a fill having a moisture content of 20 percent (wet weight) and no water table in the solid waste. In this fill the concentrations of 34 percent carbon dioxide and 7 percent methane indicated that low moisture content does not enhance methane fermentation. Carpenter and Setter also showed that maximum temperature is reached between 7 and 45 days after placement of the fill. The highest temperature readings were generally recorded near the surface. The maximum temperature at a depth of 2.10 to 4.20 m (7 to 14 ft) was about 11°C (51.8°F) less than that recorded at 0.90 m (3 ft) depth; this finding indicates that anaerobic conditions, which are generally more pronounced at greater depths, tend to decrease the maximum temperature.

Eliassen[9] found that aerobic conditions in landfills prevail near the surface and cause relatively high temperatures (62°C at a 0.9-m depth). In deeper layers, anaerobic conditions prevail and lower maximum temperatures are observed (41°C at a 3.3-m depth). Since most of the solid waste remains under anaerobic conditions, a net gas production is generally observed in landfills. Eliassen also noted that the optimum moisture content for aerobic degradation of recently deposited solid waste is between 50 and 70 percent (wet weight); in older fills the optimum is between 30 and 80 percent, an estimate based on the amount of carbon dioxide produced in laboratory studies. Optimum moisture content for anaerobic degradation processes was not determined.

Only a few studies have measured gas production rates in pilot landfills or in laboratory-scale simulated landfills in which the solid waste is enclosed to collect all the gases generated. In such studies, the rate of solid waste stabilization was reflected in the rate of gas production and in the volume of gas produced. These factors in turn were related to the moisture content and temperature of the solid waste.

Merz[10] measured the rate of gas production, gas composition, and solid waste temperatures in eight 208-liter (55-gal) drums filled with 36.7 kg (80.7 lb) (dry weight) of a composite solid waste having a dry density of 320 kg/m^3 (540 lb/yd^3) and consisting of 15 percent grass, 4 percent garbage, and 81 percent paper, with 85 percent moisture content (dry weight). The gas production occurred mainly during the first 400 days of the study. Since carbon dioxide was the only gas produced in substantial quantities and methane was detected only in trace amounts, one can conclude that refuse degradation proceeded through the acid digestion phase but not into the methanogenic phase. The volumes of gas produced ranged from 0.0016 to 0.013 m^3/kg (0.03 to 0.21 ft^3/lb) (dry) of solid waste, whereas a theoretical maximum of 0.63 m^3/kg (11 ft^3/lb) was determined by Anderson.[1] The volume was related to the garbage content of the solid waste, and examination of the waste at the end of the experiment showed that only this fraction was degraded. The grass clippings, leaves, and paper were always easily identifiable.

Merz found that gas production was related to temperature, which decreased logarithmically after reaching a maximum. Figure 15 shows this pattern for container A for the first 50 days. Insulating the container again increased gas production, but it then dropped logarithmically. Increasing room temperature by approximately 10°C (50°F) caused a third increase in gas production. A short interruption of the heating on the 230th day caused renewal of gas production, which indicates that a cold shock may enhance degradation. When the gas production data for all cylinders were related to refuse temperature, the result clearly indicates (Figure 16) that the highest gas production rates (0.1 to 0.2 m^3/ton [dry] per day) occurred within a temperature range of 30 to 40°C (80 to 100°F). This range is optimum for the proliferation of mesophilic anaerobic bacteria.[11]

Of all the factors studied by Merz,[12] moisture content was the most important. The highest gas production (13 m^3/ton) was observed in container F after it was brought to field capacity and provided with additional moisture. Moisture addition (Figure 17) decreased gas production during the first 30 days, possibly as a result of the lower temperature. In 50 to 100 days after the container was insulated, however, the addition of water generally led to increases in gas production. Several other studies also note the effects of moisture content on gas production (Figure 18). The spread of data, contradictions, and omission of important measurements with regard to the influence of temperature and moisture on solid waste decomposition indicate that further research is warranted. There was no correlation with respect to gas composition in the laboratory data collected by Merz[12] and those obtained at the Spadra pilot landfill of the County Sanitation District of Los Angeles.[13]

The highest concentration of carbon dioxide recorded in the field was 75 percent in the saturated cell, but it was as high as 98 percent in the saturated laboratory container. Field tests definitively established that methane concentrations depend upon the moisture content of the solid waste; concentrations ranged from little more than a trace in the cell to which no water was added to more than 50 percent in the cell brought to field capacity. In the laboratory study, the methane content never exceeded 1 percent and no relation to moisture content was detected.

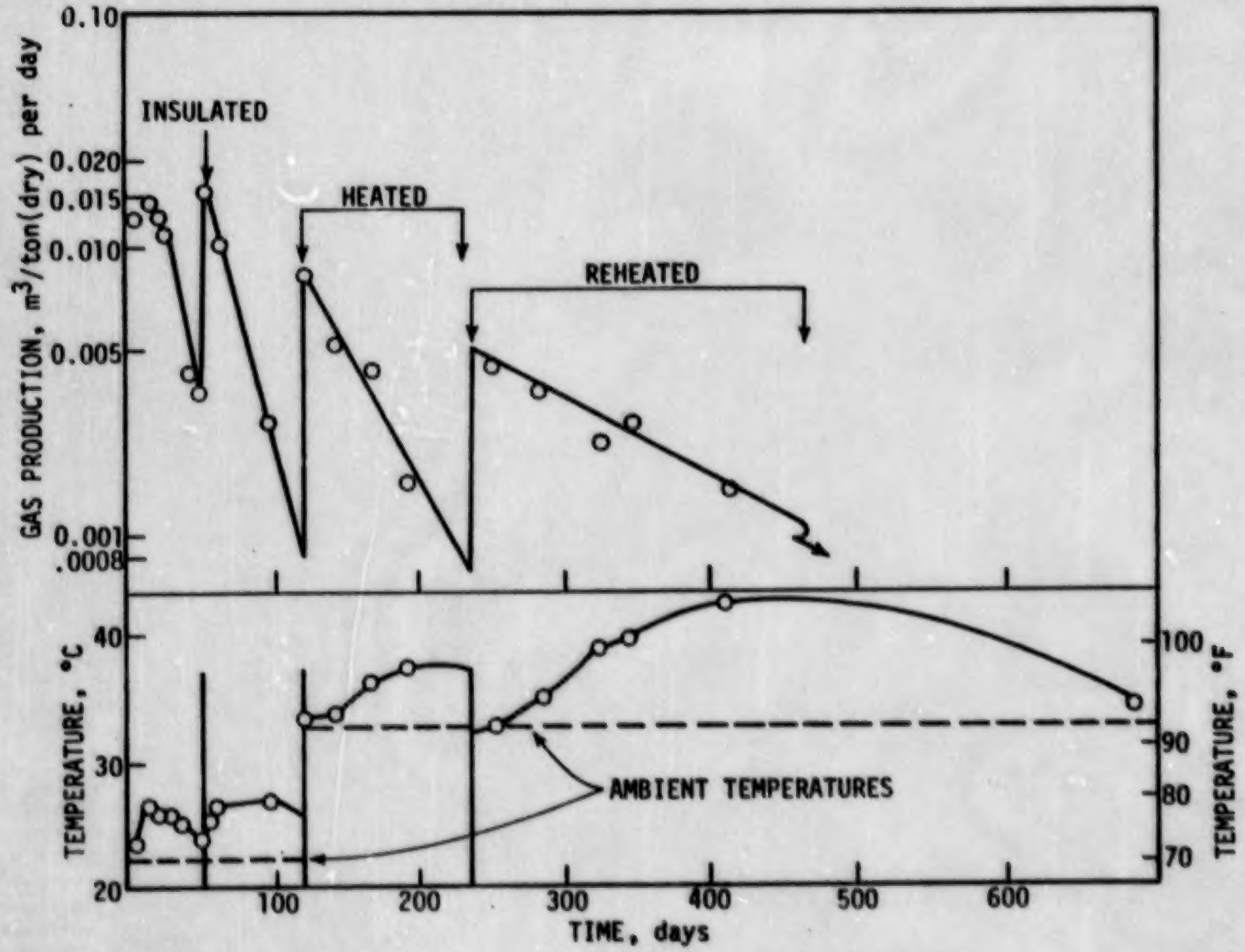

Figure 15. Influence of Temperature Changes on Gas Production Rates.[12]

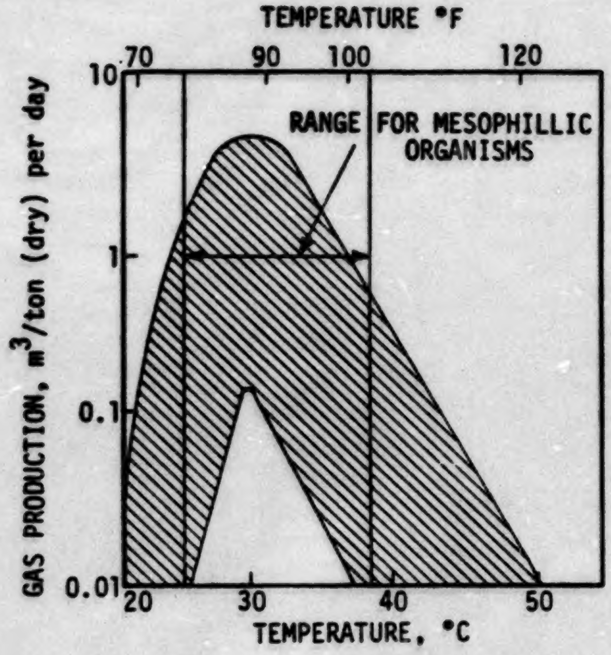

Figure 16. Optimum Temperature Range for Gas Production.[12]

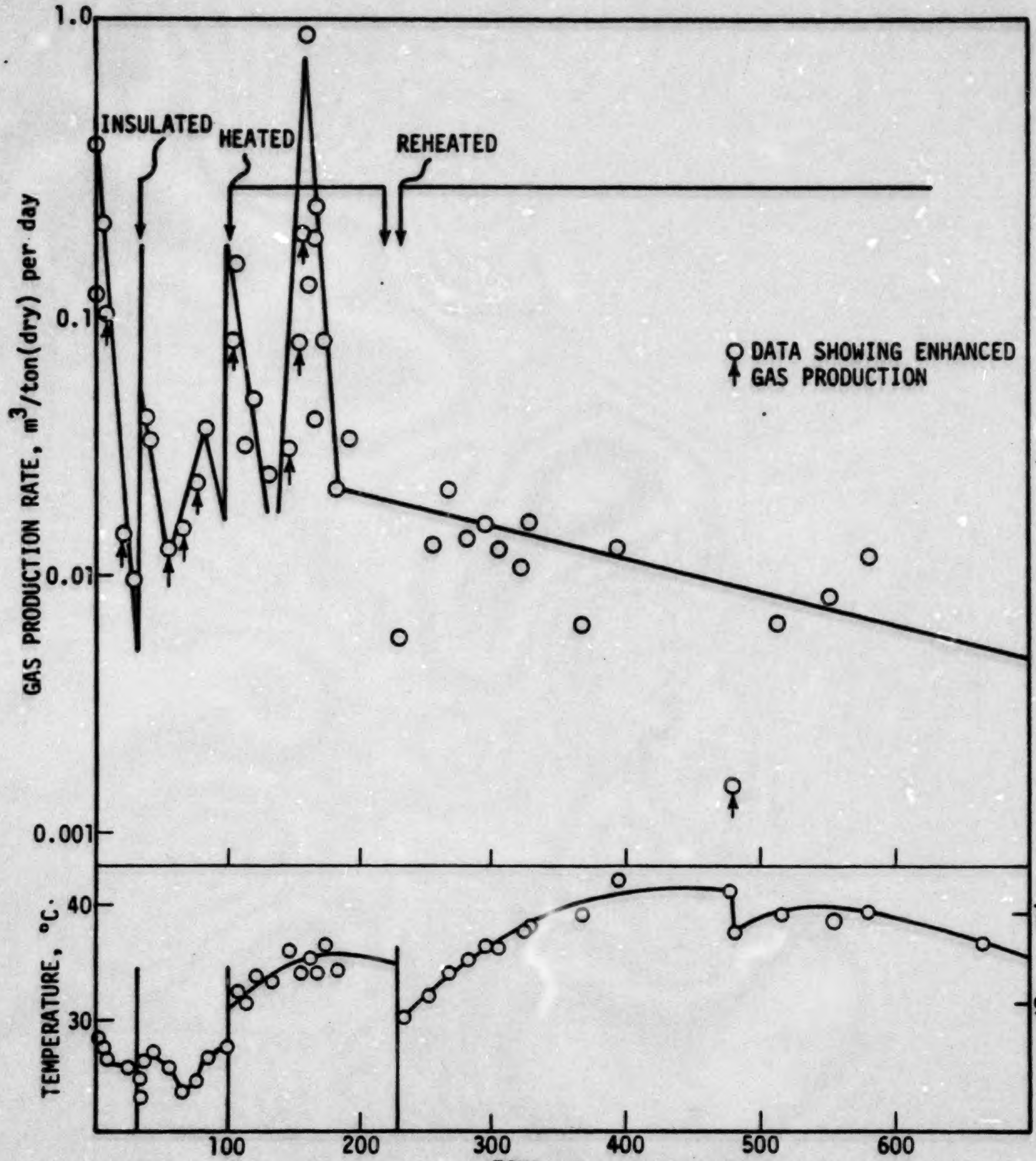

Figure 17. Influence of Moisture Addition on Gas Production Rate.[12]

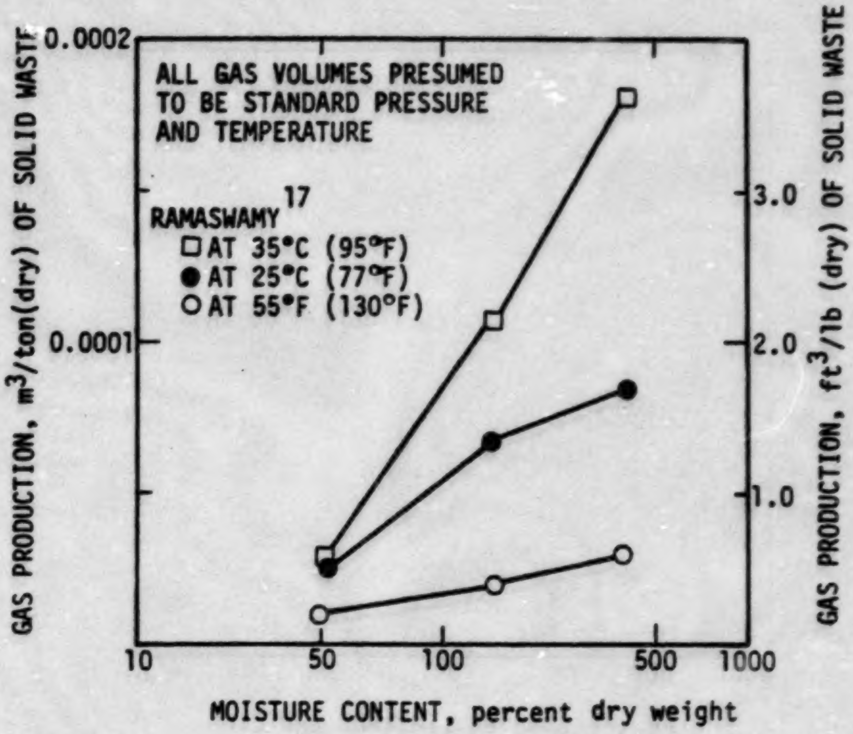

Figure 18. Effect of moisture on volume of gas produced.

To study these discrepancies further, Merz and Stone filled a (37.5 m^3 [49 yd^3]) metal container with 10,477 kg (dry) (23,097 lb) of solid waste.[13] Daily gas production (Figure 19) initially decreased logarithmically (as in the laboratory experiments) and then ceased after 50 days. Because they believed that the initial moisture content of 49.7 percent (dry weight) was below that required for optimum gas production, the investigators increased it after 60 days to 65.8 percent (dry weight). In contrast to the laboratory experiments, no immediate increase in gas production was noted. A resurgence occurred only after 250 days, when the temperature of the solid waste rose as a result of warmer weather. The gas production was highest when the temperature of the solid waste reached 46°C (115°F), possibly as a result of higher outside temperatures. This temperature is about 10°C (50°F) higher than the optimum temperature determined in the laboratory experiments. After daily gas production reached its maximum, it did not decrease logarithmically, as in the laboratory study, but rose and fell as temperature of the refuse changed. The data thus indicate that moisture content alone does not influence gas production but rather a combination of moisture and temperature. Unfortunately, no gas composition data were obtained from the pilot study. Since the total gas production was comparable to that obtained in the laboratory study, decomposition may have proceeded only through the acid fermentation phase and not into methane fermentation.

In major gas production studies conducted at the University of West Virginia at Morgantown, Dobson found that aerobic decomposition proceeded best at a moisture content of 60 percent.[14] Higher moisture contents generally favored anaerobic degradation of the solid waste, but no optimum level was found. Dobson also observed that aerobic solid waste reached a temperature as high as 57°C (134.6°F), whereas the temperature of anaerobic refuse did not increase beyond 30°C (86°F). This finding confirms the results noted by Carpenter and Setter[8] and Eliassen.[9]

Lin[15] studies gas composition in three cylinders containing shredded solid waste (cell A) unshredded solid waste (cell B) and shredded waste to which water had been added to increase its moisture content to 50 to 60 percent (wet weight, cell C). Hydrogen gas production was highest in cells A and C (19 and 20.6 percent, respectively) and lowest in cell B (5.6 percent). Maximum carbon dioxide concentration occurred after the hydrogen reached its peak and was also higher in the shredded solid waste containers than in cell B. The maximum amounts were 84.8, 55.5 and 75.8 percent in cylinders A, B, and C, respectively. Methane production was highest in cell A because of its more pronounced anaerobic environment. After 1 year, cell A contained 13 percent methane whereas cell B contained only 0.5 percent. The methane content and the ratio of methane to carbon

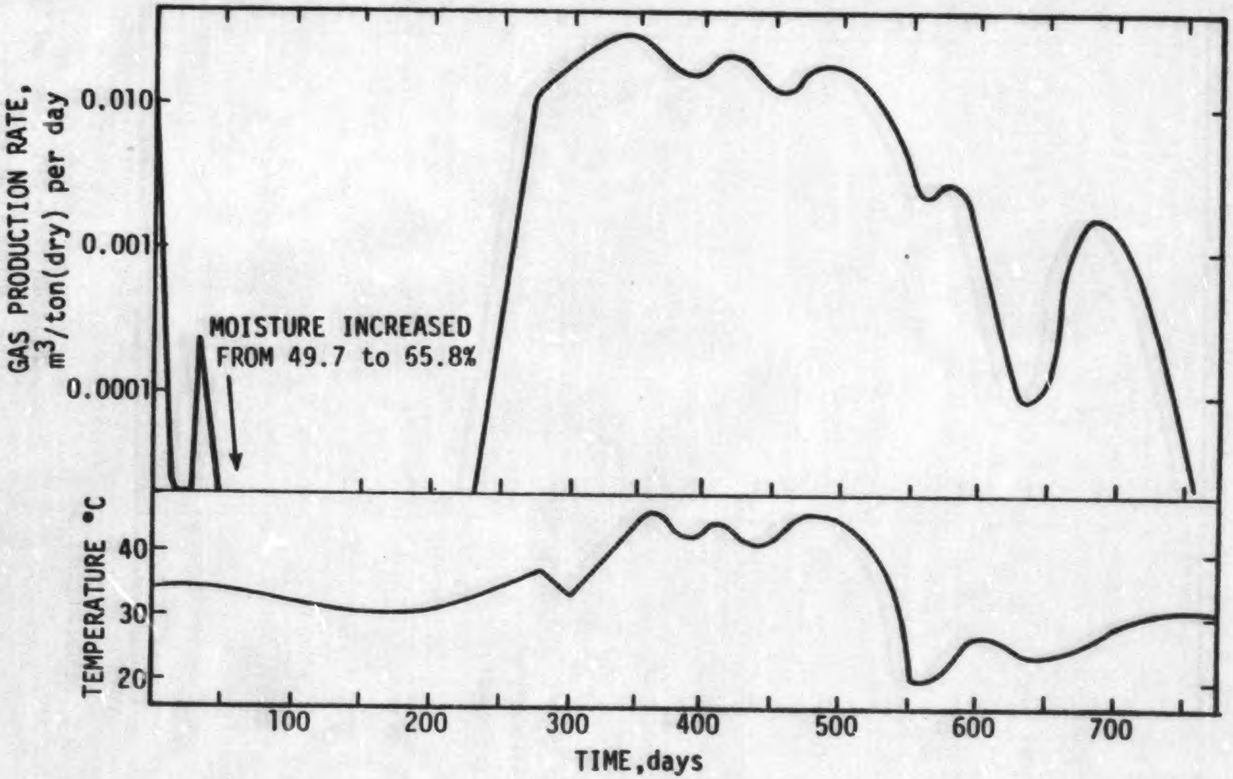

Figure 19. Gas Production Rates and Temperatures during Spadra Container Study.[13]

dioxide were generally higher in the lower refuse layers, an indication that molecular diffusion of air into the waste mass (as evidenced by the increase in nitrogen content) can inhibit methane fermentation. The high porosity of unshredded solid waste enhanced this effect. Analysis of free volatile fatty acids extracted from the solid waste with acetone showed that acetic acid was initially present in relatively high concentrations. Butyric acid became the predominant acid after 12 days.

Songanuga,[16] studying gas production in simulated solid waste, measured maximum rates of 0.234 m^3/ton per day from beef heart and 0.078 m^3/ton per day from potatoes directly after incubation. These rates are comparable to those obtained by Merz.[12] The total volume of gas produced, however, was several orders of magnitude lower, possibly an indication of major leaks in the system. The subsequent increase of the nitrogen content in the gas phase tends to confirm this. Diffusion of air into the system may have inhibited methane production, because only traces of methane were detected.

In gas production studies conducted by Ramaswamy a shredded synthetic solid waste consisted of 60 percent paper, 30 percent dog food, 7 percent leaves, and 3 percent wood.[17] Evaluation of the optimum nutrient conditions (as measured by the total gas production) showed that increasing the nitrogen content of the synthetic solid waste beyond 1.86 percent reduced the formation of fatty acids and subsequently reduced gas production. This agrees with the observation by Chian and DeWalle that a low carbon/nitrogen ratio enhances the excretion of amino acid intermediates, such as glutamic acid and tricarboxylic acid cycle intermediates (such as α ketoglutaric acid) that are less readily available for methane fermentation than acetic acid and butyric acid.[18]

Gas production and the formation of fatty acids under optimum nutrient conditions (150 percent [dry weight] moisture content and 25°C) are shown in Figure 20. Three phases are readily apparent. Phase A, which lasted for 3 days, is the aerobic-anaerobic transition phase, in which oxygen is rapidly depleted and converted into carbon dioxide. Anaerobic conditions occur when the carbon dioxide content increases beyond 20 percent. Phase B, which lasted from 3 to 13 days, is the acid fermentation stage, in which the carbohydrates are hydrolyzed into single sugars and converted into free volatile fatty acids. The carbon dioxide content of the gas increases gradually during this phase and displaces the nitrogen initially present. In Phase C (the methane fermentation stage) the excreted fatty acids are converted into carbon dioxide and methane. Phase C is characterized by a second increase in gas production and a gradual increase in the methane content of the gas. Ramaswamy found that methane fermentation becomes significant when the pH

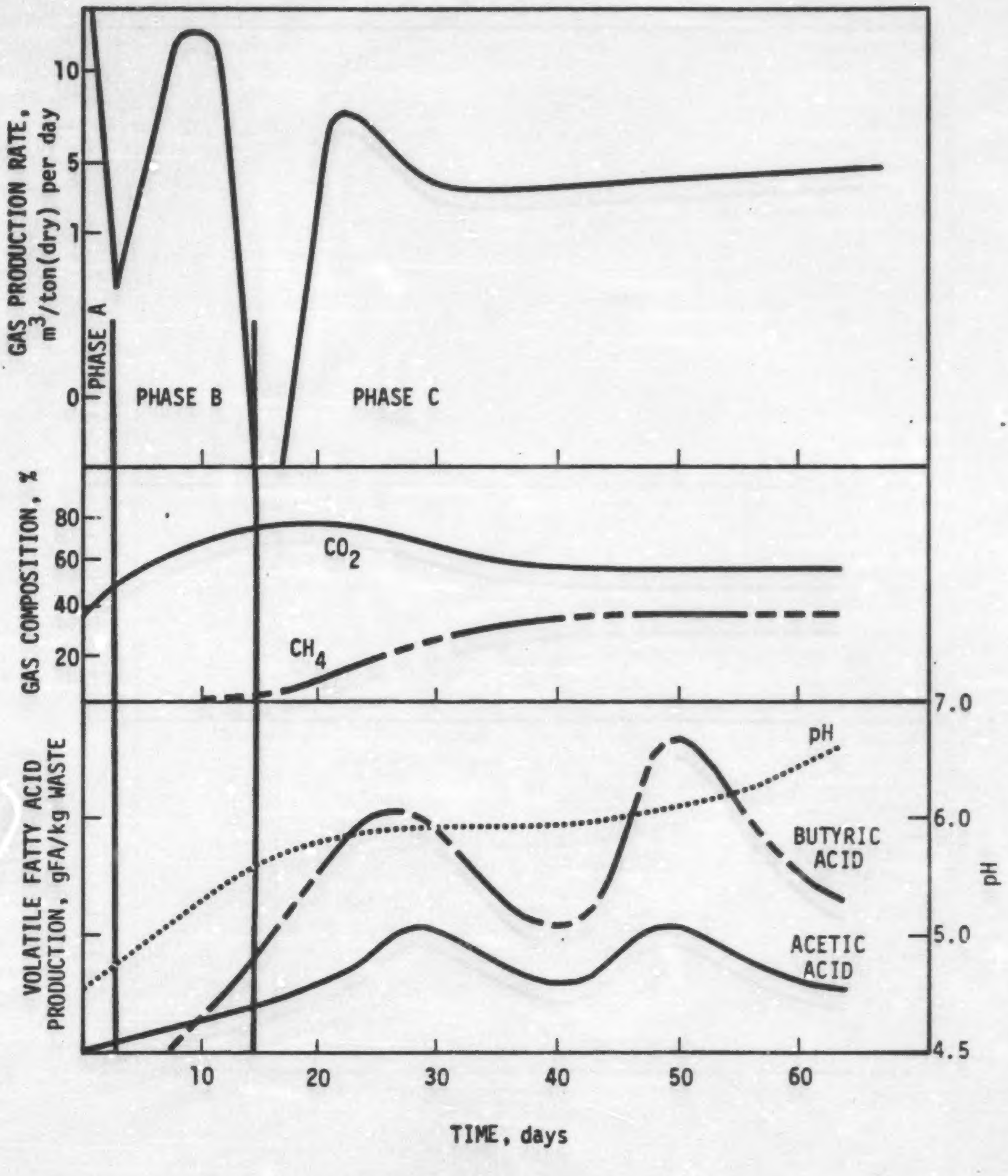

Figure 20. Gas Production and formation of fatty acids,[17] under optimum nutrient conditions.

increases beyond 4.6 to 5.4 (Figure 21). The gas composition data of Figure 20 indicate that the methane fermentation stage is not completely established after 63 days because only 36 percent of the gas consists of methane instead of 60 percent, as normally occurs in old landfills.[12] Such a high methane content will occur when the pH increases to nearly 7. Ramaswamy also found that the volume of gas produced increased at higher moisture content levels. His inability to detect a significant methane fermentation phase in solid waste maintained at 43 percent moisture (dry weight) confirmed the results of Merz,[10] who found that methane-fermenting bacteria require a moisture content of more than 100 percent (dry weight).

The gas production rates reported by Ramaswamy (approximately 1 m^3/ton per day) are one to two orders of magnitude higher than those found in previous studies and are probably due to the 30 percent food waste content, which is about 3 times greater than normal. Optimum nutrient conditions, as found in earlier studies, and shredding of solid waste will also increase the rate of stabilization.

Rovers and Farquhar[19] concurrently measured gas production, temperature, and leachate characteristics of solid waste in a test cylinder. Carbon dioxide concentrations increased during the first 20 days, but gas production did not begin until the 48th day (Figure 22) when the moisture content had risen from 98 to 133 percent (dry weight). The start of gas production corresponded to an increase in the surrounding temperature. The absence of any gas production during the first 50 days is puzzling because the carbon dioxide content of the gas was high (theoretically, gas production starts as soon as the CO_2 content exceeds 20 percent). It is possible that a gas-collection device malfunctioned. The absence of significant concentrations of methane indicated that the solid waste was still in the acid fermentation stage. Daily gas production after 50 days varied by one order of magnitude and measurements showed no consistent pattern when the moisture was increased from 133 to 194 percent (dry weight). Gas production ceased after 200 days, possibly because the acid fermentation stage had ended. A rise in pH tends to substantiate this. Total gas production (2.9 m^3/ton dry weight) is about the same as that noted by Merz and Stone[13] in the large field containers, but the moisture contents used by Rovers and Farquhar was greater. Concentrations of both organic and inorganic solids in the leachate decreased because of washout. The COD decreased faster than the fixed dissolved solids, an indication that decomposition was taking place.

Gas production rates in an actual landfill have been measured by Bishop et al.,[20] who measured gas production rates (primarily during the acid fermentation stage) in a 3315-ton (dry weight) pilot landfill. They found that an initial carbon

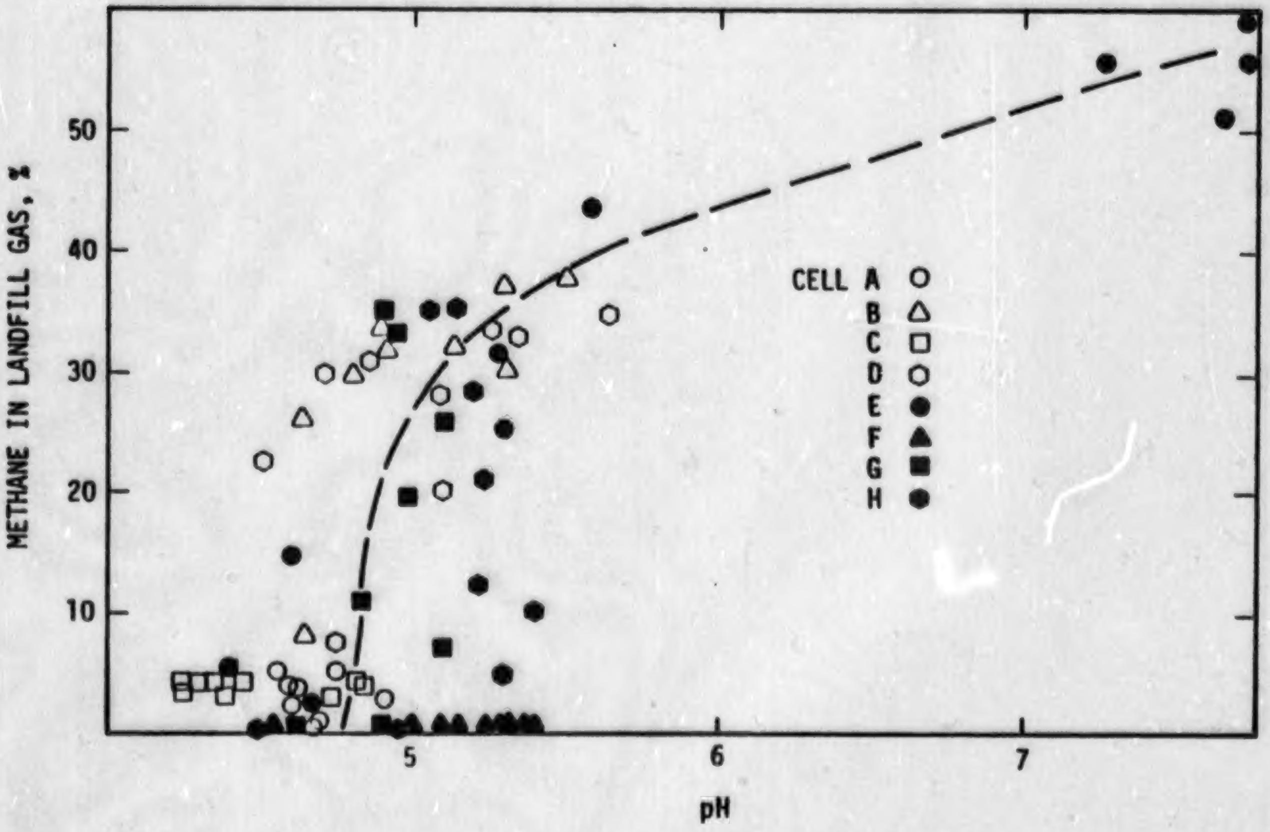

Figure 21. Relationship between methane production and leachate pH.[17]

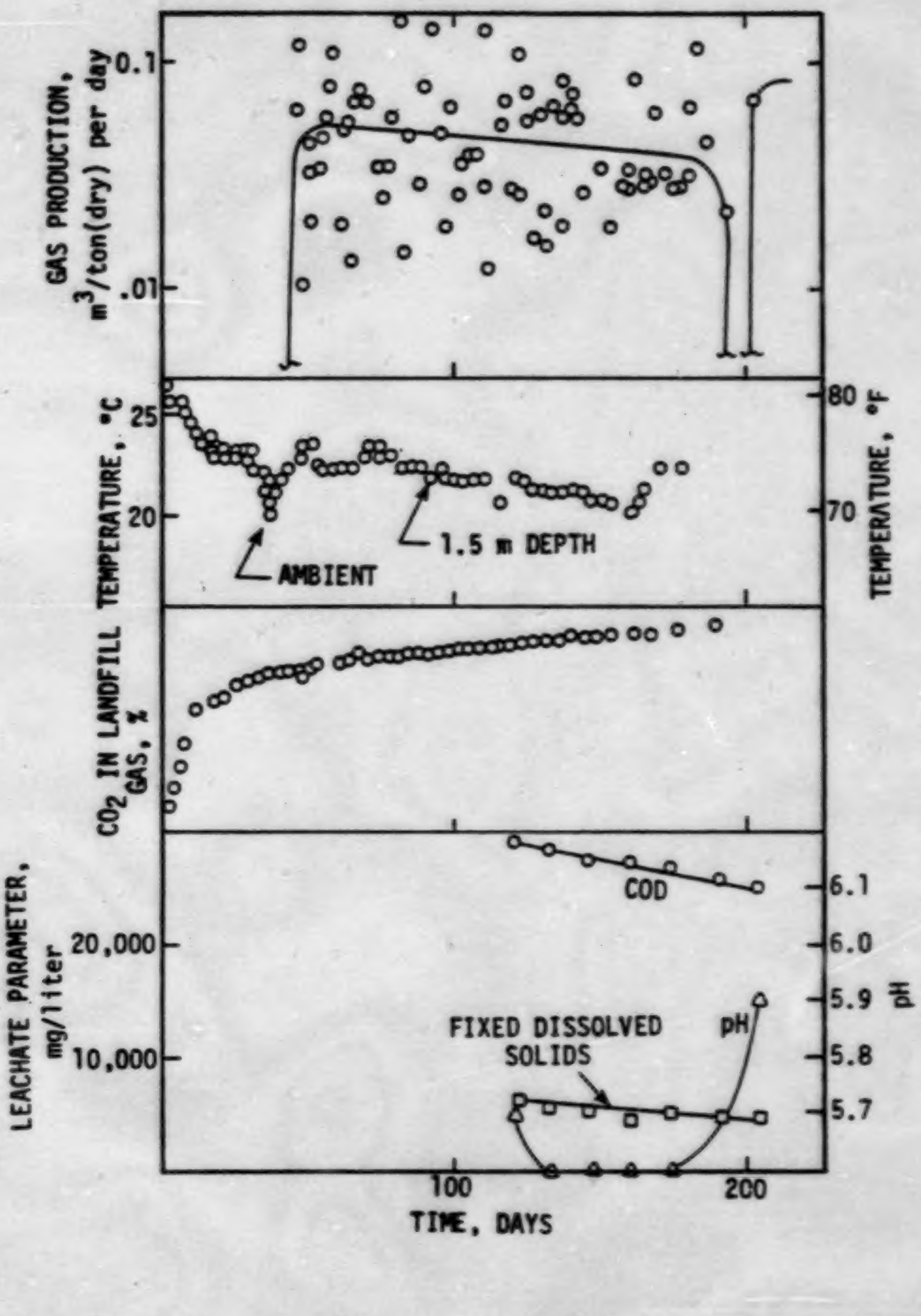

Figure 22. Gas production and leachate characteristics.[19]

dioxide bloom occurred within the first 11 days of construction and that the carbon dioxide content was as high as 80 to 90 percent. A subsequent decrease was noted as methane formed and displaced the carbon dioxide. During the 3 year observation period, methane concentrations increased to about 11 percent. An increase in moisture content as a result of rainfall entering the fill after 1 year caused a second increase in carbon dioxide and in total gas production (Figure 23). A similar phenomenom has been observed in the laboratory[10] but not in other field investigations.[13] Moisture content of the solid waste placed in the landfill was 17.4 percent (dry weight); it may have increased to 21.9 percent because 29.1 mm (1.89 in.) of rain fell during that month. In contrast to the findings of Merz and Stone,[13] gas production did not decrease at cooler temperatures. Although the rainfall entering the fill reduced the temperature, total gas production continued to rise. The measured gas production rates were approximately 4 times higher than those observed by Merz and Stone.[13]

CHARACTERISTICS

Carbon dioxide, methane, and hydrogen gas are produced in different quantities during the acid fermentation and the rate-limiting methane fermentation stages:

Acid fermentation
stage:

$$C_6H_{12}O_6 \rightarrow \underset{\text{(acetic acid)}}{2\ C_2H_4O_2} + CO_2\uparrow + CH_4\uparrow$$

$$C_6H_{12}O_6 \rightarrow \underset{\text{(acetic acid)}}{C_2H_4O_2} + \underset{\text{(propionic acid)}}{C_3H_6O_2} + CO_2\uparrow + H_2\uparrow$$

$$C_6H_{12}O_6 \rightarrow \underset{\text{(butyric acid)}}{C_4H_8O_2} + 2\ CO_2\uparrow + 2\ H_2\uparrow$$

Methane fermentation
stage:

$$\underset{\text{(acetic acid)}}{C_2H_4O_2} \rightarrow CO_2\uparrow + CH_4\uparrow$$

$$\underset{\text{(propionic acid)}}{4\ C_3H_6O_2} \rightarrow 2\ H_2O + 5\ CO_2\uparrow + 7\ CH_4\uparrow$$

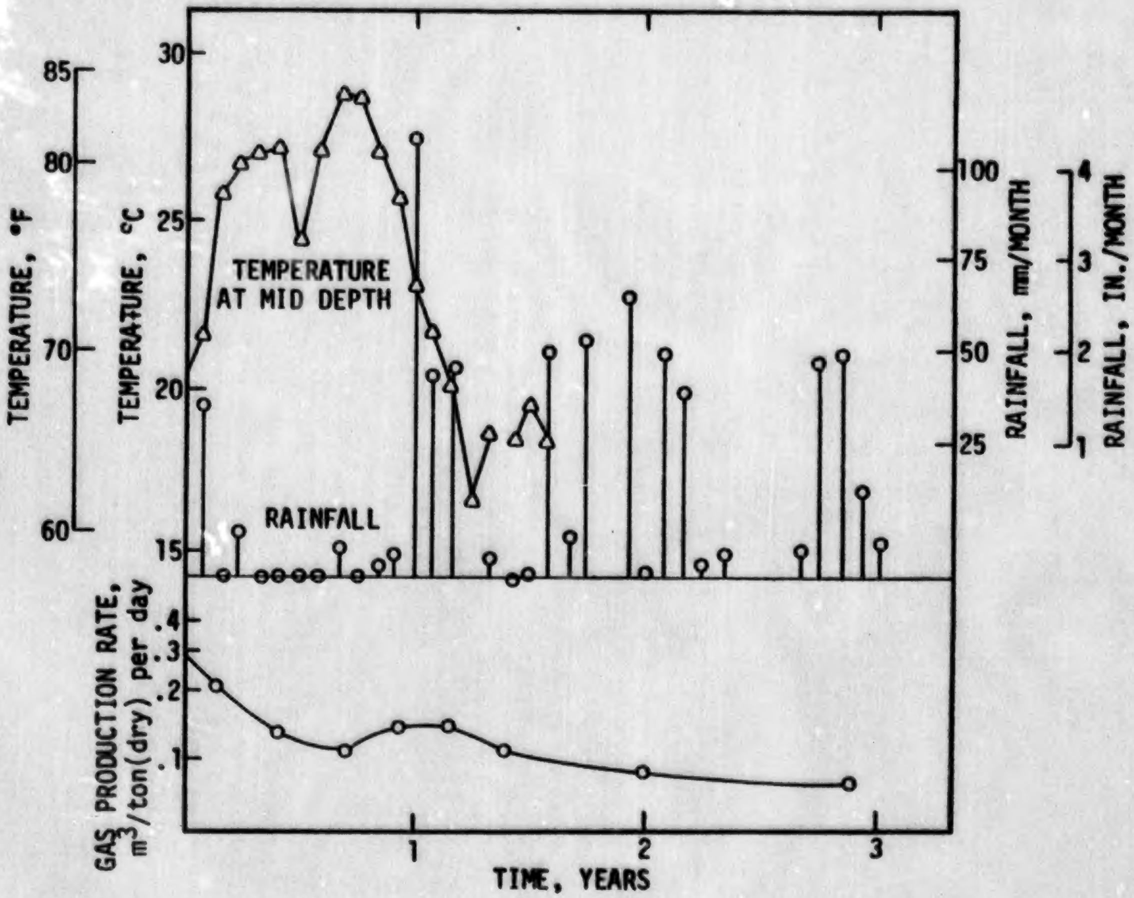

Figure 23. Gas production at Azusa landfill.[20]

In the first stage, carbohydrates are converted into acetic and butyric acid and, to a lesser extent, into propionic acid. The gases produced are primarily carbon dioxide and small amounts of hydrogen gas. In the methane fermentation stage, the fatty acids are transformed into equal amounts of methane and carbon dioxide. Because carbon dioxide is soluble in water, the actual gas composition is generally 65 percent methane and 35 percent carbon dioxide.

When fatty materials, such as stearic acids, are degraded aerobically there is even a negative production of gas

$$C_{18}H_{36}O_2 + 26\ O_2 \rightarrow 18\ CO_2 + 18\ H_2O$$

whereas under anaerobic conditions there is again significant gas production:

$$C_{18}H_{36}O_2 + 8\ H_2O \rightarrow 5\ CO_2\uparrow \quad 13\ CH_4\uparrow$$

Reported gas composition data are summarized in Table 19, in which the most important result is the effect of moisture on the intensity of fermentation. High rates of decomposition do not occur unless the weight of water in the fill is approximately equal to the weights of the refuse and soil. This is shown both by the increasing methane content of the gas and its decreasing nitrogen content. The nitrogen content is sharply reduced at high gas production rates because it is physically flushed out of the voids, as illustrated in Table 19 with regard to the Spadra study. High nitrogen contents in gas samples are indicative of low gas production rates.

No carbon monoxide has been detected in landfill gas analyses; this is not surprising in view of the biochemistry of the relevant fermentations. Hydrogen sulfide is found in trace amounts, the quantity depending upon the quantity of municipal solid waste and night-soil in the waste. Other than methane, gaseous hydrocarbons are not produced in the fermentations.

The only significant gases produced by landfills are carbon dioxide and methane. The small amounts of hydrogen found on occasion are indicative of cellulose decomposition. The composition of the gas generated in landfills is highly unpredictable because of the variations in waste composition, particularly water, garbage, and sewage solids. Peak rates of gas production and highest methane concentrations occur in wet fills containing large quantities of garbage.

TABLE 19. SUMMARY OF LANDFILL GAS COMPOSITION[a] (percent by volume)

Source	CH_4	CO_2	H_2	H_2S	CO	N_2	O_2	Reference
Seattle	60	40						21
San Francisco	50	25	8			6	0	21
New York	45	30	0.1			22	3	21
Arlington	43	32				17		21
Los Angeles								
Branford	46	40						21
Holly (2½ yr)	43	32				17		21
Long Beach								
Spring St. (20 yr)		2					19	21
Dominques (8 yr)		11					7	
Paramount (5 yr)		27					3	
Riverside								
(2-4 mo)	5	36				57	1	21
(3 yr)	6	18				20	1	
(4 yr)	65	31				3	0.8	
(4-5 yr)	33	29				38	1	
(5-6 yr)	11	18				71	0.5	
Test bin (damp)								
(1 mo)	0	4				83	13	
(2 mo)	25	35				38	2	
(4 mo)	46	38				15	0.5	
Test bin (dry)								
(1 mo)	0	14				82	4	
(2 mo)	9	10				73	8	
(4 mo)	3	5				77	15	

TABLE 19 (continued)

Source	CH_4	CO_2	H_2	H_2S	CO	N_2	O_2	Reference
Azusa	9	28	1		0		0.5	22
Kennett								
(6 mo)	0.5	22		0	0		4	23
(1 yr)	0.5	25		0	0		1	
(2 yr)	1	20		0	0		4	
Spadra								
23% water[b]								24
(6 mo)	0.2	25	1	T		73	0	
(1 yr)	0.5	23	3	T		74	0	
(2 yr)	1	26	0.5	T		71	0	
30% water								
(6 mo)	1	37	2	T		59	0.1	
(1 yr)	7	48	0.6	T		45	2	
(2 yr)	19	38	0	T		43	0.6	
44% water								
(6 mo)	32	64	3	T		0.6	0.2	
(1 yr)	48	48	4	T		0	0	
(2 yr)	54	46	0	T		0	0	

[a] Number in parenthesis indicate age of fill at time of sampling.

[b] Moisture content on percent wet weight basis.

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SECTION 8

GAS MIGRATION AND EFFECTS

MECHANISMS

Gases move through soil because of pressure and diffusion. A net component of gas moves in the direction of decreasing pressure, and the diffusion is caused by gradients in gas concentrations. In mixed gas systems, each gas flows in its own concentration gradient; the various gases therefore may flow in different directions. Consider a simple case of a landfill gas composed of methane and carbon dioxide flowing into an ambient gas composed of nitrogen and oxygen. All four gases will have a net component of flow away from the landfill because of the total pressure gradient caused by generation of gas in the landfill. Methane and carbon dioxide will also diffuse away from the landfill because of concentration gradients, while nitrogen and oxygen will do the opposite. Both pressure flow and diffusion flow will force the landfill gases (methane and carbon dioxide) away from the landfill, although total gradients would be expected to dissipate rapidly.

As the gas moves through the soil surrounding the landfill, it may be adsorbed onto the soil particles. Carbon dioxide, which is particularly susceptible to adsorption, may react with the soils and release solubilized materials, such as calcium and magnesium.

ANALYTICAL REPRESENTATION

Early attempts to model gas flow surrounding sanitary landfills were reported by the California State Water Pollution Control Board.[1]-[3] The authors delineated a pressure flow governed by Darcy's law and a diffusion flow governed by Fick's law. The investigators recognized that gas flow around sanitary landfills is a mixture of different gas types, but their analyses accounted for this only indirectly.

Moore discusses in detail the theory of gas movement in porous media.[4] The discussion deals with diffusional flow in a capillary tube, pressure flow in a capillary tube, combined pressure and diffusional flow, and flow with chemical reaction.

He also presents an equation describing flow of gases around sanitary landfills. A flux equation must be written for each component:

$$-N^{O}A = \frac{1}{\tau} \sum_{r_{min}}^{r_{max}} f_v(r) \left\{ \frac{\frac{\bar{D}_{AB}}{RT} \cdot \frac{\partial C_A}{\partial Z}}{C - (1 - \sqrt{\frac{M_A}{M_B}}) C_A + \frac{D_{AB}}{RTK_A r}} + \frac{C_A}{4} \left[\frac{RTr^2}{2\mu} + \frac{\pi K_{Ar}}{C_A + C_B \sqrt{\frac{M_B}{M_A}}} \right] \frac{\partial C}{\partial Z} \right\}$$

where $N^{O}A$ = total (pressure plus diffusional) flux of gas A relative to fixed coordinates

τ = tortuosity factor

r = pore radius

r_{min} r_{max} = limits on pore size distribution approximation

$f_v(r)$ = volume fraction having pore radius, r

D_{AB} = (constant) diffusion coefficient for gases A and B

R = gas constant

T = temperature

C_i = concentration

Z = spatial distance

C = total gas concentration

M_i = molecular weight of gas i

K_A = (constant) Knudsen diffusion coefficient for gas A

μ = gas viscosity

In addition, conservation of matter must be reflected in continuity equations:

$$\frac{\partial C_A}{\partial t} = - \frac{\partial N_A}{\partial z} + S_A$$

where S_A = amount of component A created (or destroyed).

Note that in these equations the properties of the gases and of the porous medium are decoupled:

gas properties: D_{AB}, M_A, M_B, K_A, μ.

porous medium properties: r, r_{min}, r_{max}, $f_v(r)$.

This mathematical analogue describes the flow of gases around sanitary landfills. Because of the many variables a relatively large computer is required for computation.

In a description of design criteria for gas migration control devices Moore and Rai[5] discuss the advantage of computer simulation of gas migration, both with and without control devices. They also present a typical design situation involving migration of methane from a landfill in which control measures are evaluated by computer simulation.

Moore and Jayco[6] discuss the development of design charts to be used in predicting time-dependent concentrations of gases at any position around a sanitary landfill. Through the use of these charts the effects of carbon dioxide on groundwater quality and potential gas hazards involving methane can be assessed. The charts allow prediction of the maximum distance of gas excursion and the time required for maximum excursion to occur. The charts are based on relatively simple landfill site geometry and flow boundary conditions.

INCIDENTS OF GAS MOVEMENT

Several field studies of the flow of gases in and around sanitary landfills have been done to determine the quantities of gases produced and to monitor their concentrations as a function of time and their positions in and near landfill sites.

The California State Water Quality Control Board[2,3] described the results of a study conducted at a gravel pit in an alluvial fan formed from outwash of the San Gabriel River near Azuza. The site was a 0.3 hacter (3/4-acre) pocket in a larger gravel pit. Three walls of the pocket were 8 meter (25 feet) high. The pit was filled with over 4000 tons of municipal refuse, which was placed in 1.8 m (6 ft) lifts separated by 150 and 250 mm (6 and 10 in) of silt and topped by a 400-mm (16 in) layer of silt. The investigators installed gas sampling probes and temperature-moisture probes. Over a period of 4 years after construction of the landfill the gases were analyzed for carbon dioxide, oxygen, hydrogen, carbon monoxide, and methane.

The principal gases detected were methane and carbon dioxide. Figures 24 and 25 indicate the variation in these gases with time and position. The investigators presented calculations indicating that about 560 ton/ha per year of carbon

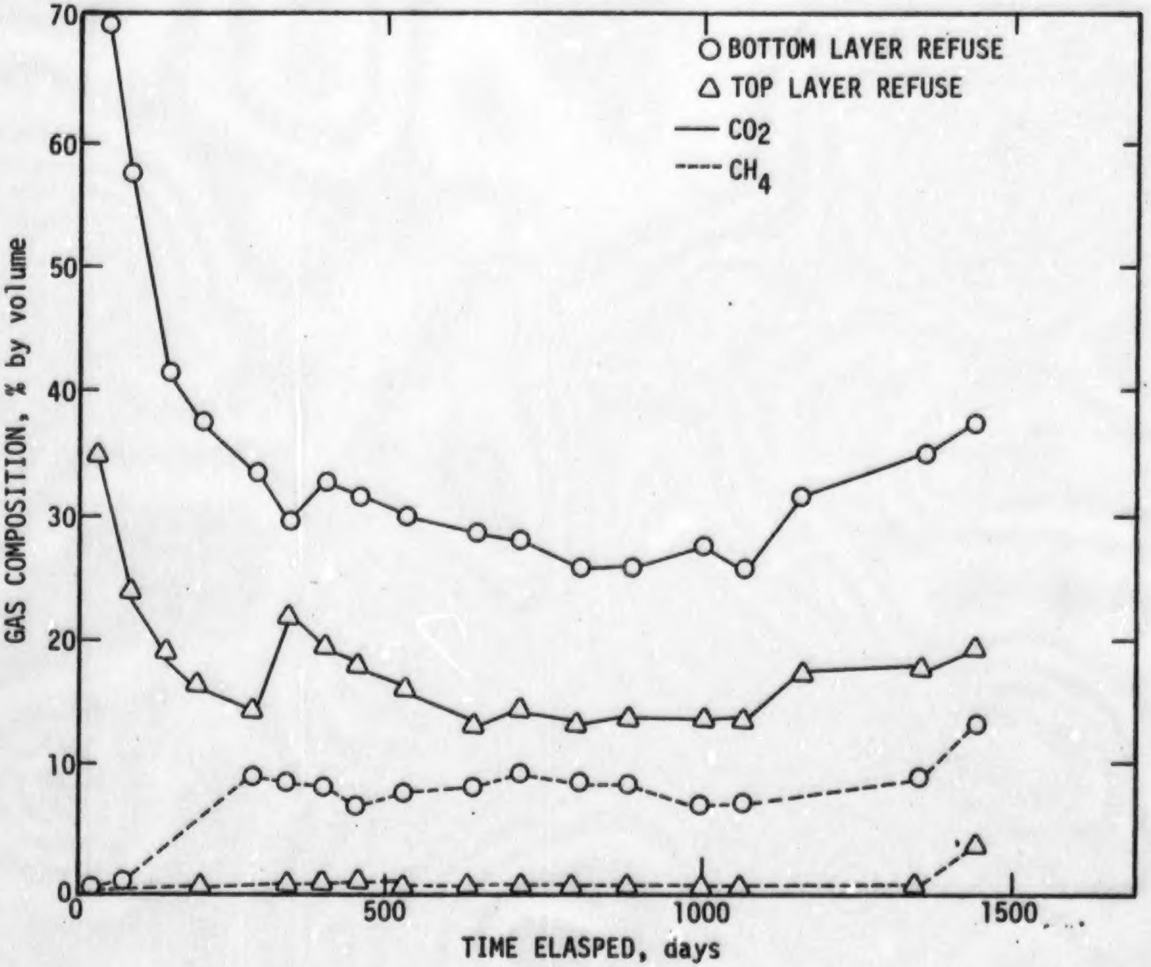

Figure 24. Variation in Gas Composition (at Azusa Test Site) with Time.[3]

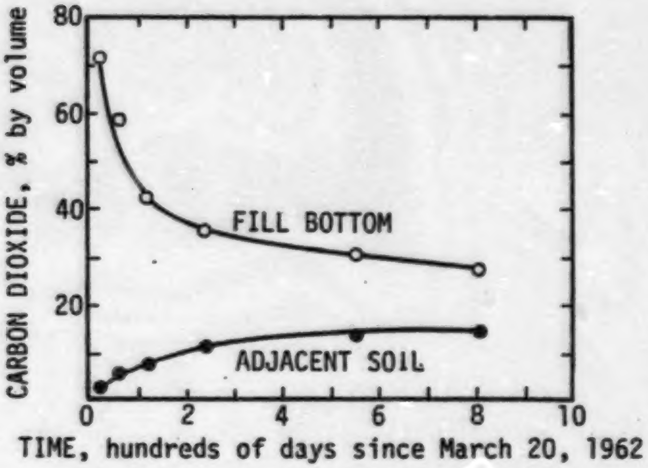

Figure 25. CO_2 in Soil Atmosphere under and Adjacent to[2] the Azusa Landfill.

dioxide and 56 ton/ha per year of methane left the landfill in the initial stages of decomposition. They determined approximately 1/24 of the gas remains in the ground surrounding the landfill, and the rest escapes into the atmosphere.

These studies were supplemented by data from two other sites. The Los Angeles County Calabasas landfill at Agoura was provided with instrumented wells placed 15.2 m (50 ft) and 30.4 m (100 ft) from the landfill, which is in alluvium carbon dioxide concentrations, which increased from 15 to 19 percent by volume over a 3-month period, indicated that gas was moving away from the landfill.[3]

In a similar program, three wells (each containing sampling probes at 7.6 [25], 15.2 [50], 22.8 [75], and 30.4 [100] m [ft] depths) were installed 9.1 [30], 14.5 [48], and 28.2 [93] m [ft] from the Palos Verdes landfill at Rolling Hills, California.[3] Table 20 summarizes the carbon dioxide concentrations, which ranged from 1.8 to 15 percent by volume; the peak level of carbon dioxide movement occurred at the 30.5 m (100 ft) depth.

In an investigation[3] of methane intrusion into a California community, 24 gas-sampling probes were installed 0.61 m (2 ft) below the surface approximately 244 m (805 ft) from the landfill, as shown in Figure 26. The gas concentrations are given in Table 21. The landfill was located in a region composed of sand and gravel overlain by clay and loam. The maximum excursion of the 5 percent methane contour was approximately 137 m (452 ft) from the edge of the landfill.

Gas measurements made in tests under and near a sanitary landfill at Monrovia, California are given in Table 22.[1] Test hole No. 2 was under the fill, No. 1 was within a few meters of the fill, and No. 3 was approximately 210 m (693 ft) from the edge of the fill. Concentration of carbon dioxide and methane ranged as high as 45 percent by volume. The data indicate the significant effect on soil atmosphere caused by migrating landfill gases.

An annotated bibliography on landfill gas[7] describes reported incidents of gas migration and the resulting adverse effects. Two incidents of loss of human life are cited. Other, less dramatic incidents include construction problems and the forced vacating of buildings because of odorous and explosive gases. Migrating gas has also stunted and killed vegetation, but botanical gardens and shopping centers have been constructed on completed landfills where migrating gases have been controlled.

TABLE 20. CARBON DIOXIDE CONTENT OF SOIL ATMOSPHERE AT PALOS VERDES LANDFILL[3,a] (percent by volume)

Sampling[b] depth, m	May 25			June 8			June 23			July 21			August 25		
	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C
30.5		11.2	8.4		11.0	14.4		9.8	11.6		10.8	13.8		11.0	15.0
22.9	5.8	3.8	5.0	6.4	4.8	7.0	6.6	3.2	5.4	6.2	5.0	7.0	7.2	5.6	7.2
15.2	7.0	2.8	3.0	6.4	2.6	3.0	6.6	3.0	2.6	7.2	3.2	3.4	6.8	3.4	3.6
7.6	9.0	3.2	1.8	7.6	2.6	1.8	9.0	3.0	1.8	8.0	3.6	2.0	8.0	3.6	1.8

[a] Sampling conducted in 1965. Wells located in line perpendicular to edge of fill at distances as follows: A, 9.1 m; B, 14.5 m; C, 28.2 m.

[b] 1 m = 3.3 ft.

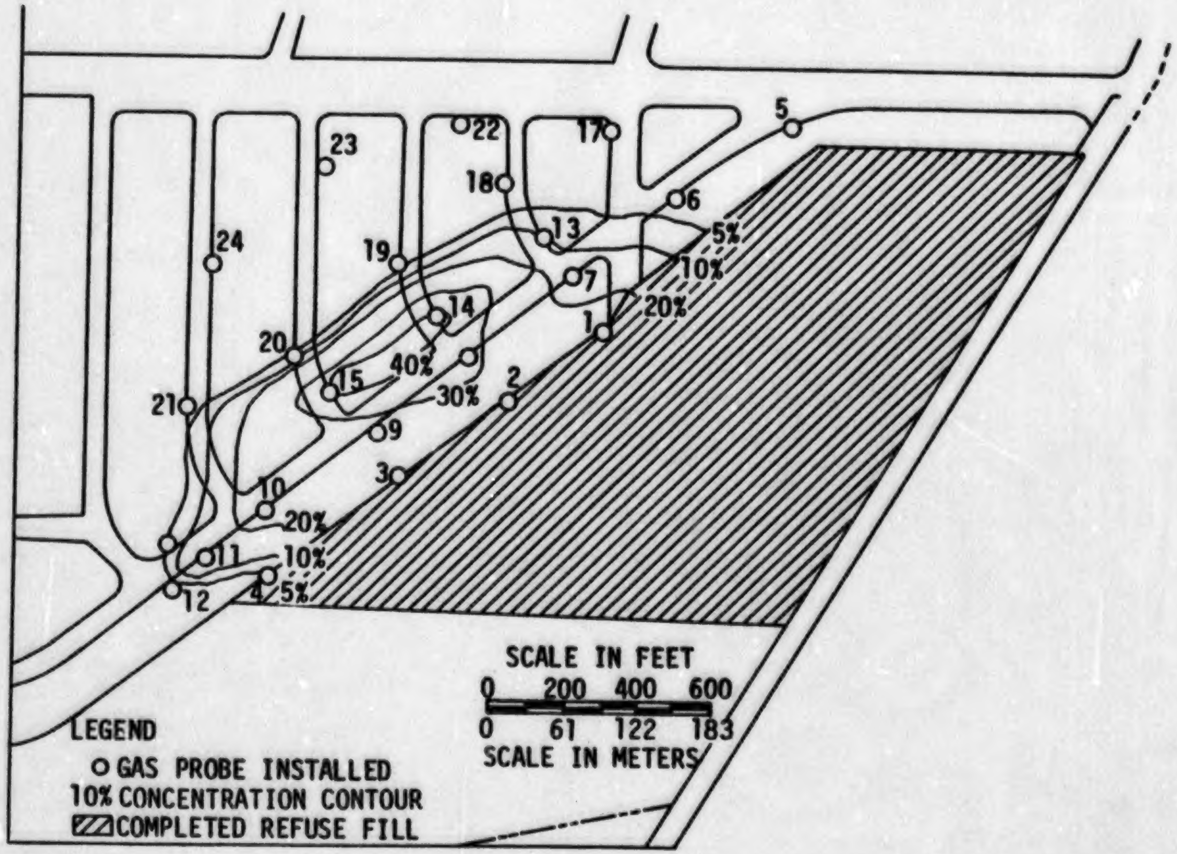
5
22
17
23
18
6
13
5%
24
19
10%
7
14
20%
1
20
15
40%
2
30%
21
9
10
3
20%
11
10%
12
4
5%
SCALE IN FEET
0 200 400 600
0 61 122 183
SCALE IN METERS
LEGEND
○ GAS PROBE INSTALLED
10% CONCENTRATION CONTOUR
▨ COMPLETED REFUSE FILL

Figure 26. Methane in soil adjacent to a landfill.[3]

TABLE 21. GAS ANALYSES IN THE VICINITY OF A LANDFILL[3] (percent by volume)

Probe no.	CH_4	O_2	CO_2	N_2
1	28.85	2.75	33.50	34.90
2	27.70	0.63	42.10	29.57
3				
4	5.31	2.95	25.70	66.04
5	0.00	23.02	2.56	74.42
6	0.14	10.48	12.40	76.98
7	14.15	1.80	30.46	53.59
8	32.20	0.66	30.90	36.24
9	26.60	1.11	39.80	32.49
10	26.50	24.82	27.20	21.48
11	34.48	0.68	22.40	42.44
12	0.00	20.43	1.56	78.01
13	8.74	5.33	23.75	72.18
14	42.10	0.29	42.40	15.21
15	43.40	0.49	29.30	26.81
16	4.95	15.35	4.73	74.97
17	0.00	19.70	3.89	76.41
18	1.21	14.99	3.86	79.94
19	0.00	7.27	6.24	86.49
20	0.00	15.11	6.20	78.69
21	0.00	16.93	2.78	80.29
22	0.00	16.11	3.70	80.19
23	0.00	17.20	3.08	79.72

TABLE 22. COMPOSITION OF GAS AT MONROVIA TEST HOLES[1]

Test hole no.	Depth, m (ft)	Gas composition, vol %			
		CO_2	CH_4	O_2	N_2
1	6.1 (20)	28.6	17.6	1.8	52.0
	12.2 (40)	26.2	12.9	2.8	58.1
	18.3 (60)	29.8	10.6	1.2	58.4
	24.4 (80)	32.2	14.0	1.1	52.7
	30.5 (100)	12.6	6.5	14.8	66.1
2	6.1 (20)	43.5	44.9	1.7	9.9
	18.3 (60)	31.4	14.7	3.8	50.2
	24.4 (80)	45.3	21.9	1.9	30.9
	29.3 (96)	40.8	22.2	2.7	34.3
3	6.1 (20)	0.03	0.0	21.5	78.5
	12.2 (40)	9.5	0.0	9.7	80.8
	18.3 (60)	15.4	0.0	6.4	78.2
	24.4 (80)	17.0	0.2	4.1	78.7
	26.2 (86)	23.7	1.3	1.5	73.5

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SECTION 9

GAS CONTROL METHODS

Gas produced in a sanitary landfill can migrate from the site and accumulate in or beneath dwellings or other structures. Accumulations of these gases can cause offensive odors, vegetation kills, and fire or explosion. Methane in 5 to 15 percent concentrations in the landfill gas is highly explosive in the presence of oxygen. Gas control is therefore an important consideration in design and operation of a sanitary landfill. When the landfill cover soil is purposely made relatively impermeable to limit infiltration, it will hinder vertical escape of gas from the covered waste and promote horizontal or lateral migration away from the fill. Vents and barriers are the principal means of controlling gas migration and thus protecting nearby structures and vegetation. Control by recovery and utilization of the gas is sometimes feasible, and this option is being investigated. Both aspects of controlling landfill gases are considered in the following subsections.

GAS CONTROL DEVICES

Gas control devices may be categorized into six types: 1) pipe vents with natural convection, 2) pipe vents with forced convection, 3) trench vents with natural convection, 4) trench vents with forced convection, 5) impervious trench barriers and 6) hybrid trench systems.[1] Each of these devices can alleviate or control gas migration.

Pipe vents can relieve localized gas accumulations in a sanitary landfill.[1] Normally some type of convective flow must be present or must be induced if pipe vents are to be effective. A series of pipe vents surrounding a landfill will naturally approach the results obtainable with a trench as the number of pipe vents is increased.

Trench vents consist of trenches cut around landfills and filled with coarse gravel. A trench vent may be designed to vent naturally to the atmosphere or to be pumped in the exhaust or recharge mode. Trenches with natural convection are not generally effective.[2] Trenches with forced convection can alleviate and control gas migration, most effectively with recharge pumping.[2]

Recharge pumping is effective because of the positive pressure barrier that forces the gas back toward the landfill. Problems may arise when the gas has already migrated beyond the point at which a pumped recharge trench vent is installed; the gas down gradient of the trench will be forced farther from the fill. Exhaust pumping is effective because it attracts gas from the fill to the vent for release to the atmosphere, incineration, or methane recovery.

Impervious trench barriers can alleviate and control gas migration if they extend downward to an impervious formation. These are the most economical barriers to construct and maintain when the impervious formation is shallow.

The least expensive, and often acceptable, barrier material is the soil at or near the landfill site. The properties of soil can be changed substantially by simple techniques, such as manipulation of moisture content and bulk density. Generally, clay soils are preferred. Less desirable soils can be improved by mixing them with commercially available sealants, such as bentonite clay and cement, to reduce their permeability. Moisture content can severely affect soil properties, including gas permeability. Unless the moisture content is regulated, shrinkage of clay soils can cause cracks through which significant amounts of water and gas can escape.

A hybrid trench system consists of a trench with natural or forced convection, backed by an impervious barrier. A hybrid system can alleviate and control gas migration when used with high recharge pumping. Such systems are usually less effective than forced convection trenches alone.

Monitoring of the concentration and movement of gases is essential to determine the effectiveness of the control system and to ensure that the control does not cause alterations that produce high concentrations of gas.

GAS RECOVERY AND UTILIZATION

The growing need to control the migration and adverse effects of landfill gas has given impetus to schemes for gas recovery and utilization. Recent work has shown that recovery and utilization of landfill gas is both technically and economically feasible under favorable conditions of gas generation, location, and demand. Recovery and utilization systems that are competitive with most other waste-to-energy processes should be considered in solid waste resource recovery programs.

Potential uses of landfill gases are as a fuel gas at various quality levels and as a process feed stock. Table 23 lists

TABLE 23. APPLICATIONS FOR LANDFILL GAS AS FUEL[a]

Application	Processing	Higher heating value (HHV)[b] kJ/std m^3 (Btu/scf)	Limitations
Direct fuel	Condensate and particulate removal	17167 to 18387 (460 to 490)	Must be consumed at landfill
Direct fuel	Dehydration	17167 to 18287 (460 to 490)	Can be transported moderate distances by pipeline
Direct fuel	Dehydration and partial carbon dioxide removal	24258 to 27990 (650 to 750)	Can be transported moderate distances and mixed with natural gas at low ratios
Direct fuel	Dehydration and total carbon dioxide removal	35080 to 35827[c] (940 to 960)	Can be mixed with natural gas at intermediate to high ratios
Direct fuel	Dehydration, carbon dioxide and nitrogen removal	35827 to 36947 (960 to 990)[b]	Can be mixed with natural gas at intermediate to high ratios
Direct fuel	Dehydration and carbon dioxide removal plus sweetening with approximately 1 percent propane	37320+ (1,000+)	Equivalent to natural gas

[a] Reference 3.

[b] 1 percent other hydrocarbons add about 1120 kJ/std m^3 (30 Btu/scf) to HHV.

[c] Palos Verdes landfill molecular sieve product, during 204 days of operation, averaged 99 percent methane, HHV = 36947 kJ/std m^3 (990 Btu/scf).

applications of landfill gas as a fuel at various quality levels.

Conversion of methanol is a representative process in which landfill gas is used as a feed stock. Other potential conversion products include ammonia and urea. Of these three possibilities methanol conversion is perhaps the most feasible.

Raw landfill gas can be consumed on or immediately adjacent to the landfill site to generate steam for process or heating use, or to generate electricity via a gas engine, gas turbine, or steam turbine. The advantages of onsite utilization is that little or no processing is required.

Because gas recovery is likely to be economical for only a portion of the time gas is generated in the landfill, it is desirable to maximize gas production during the period of recovery. Another major concern is the effects of time on gas recovery rate and on stability of composition, since the marketability of landfill gas in any form depends upon meeting customer specifications for quality and demand over a relatively long period. Unless the producer can provide assurance of an adequate supply of landfill gas over an adequate time period, the high amortization rate may make use of this gas uneconomical or noncompetitive with other fossil fuels or waste-to-energy processes.

Table 24 shows the minimum landfill gas flows needed for economic viability in applications by private enterprise and local government of the landfill gas uses studied by Lockman and Associates. When the practical aspects of recovery of landfill gas are taken into account (recovery efficiency, boundary losses, collection losses, and gas generation before and after recovery is in operation, etc.), current estimates of gas recovery range from 0.013 to 0.047 standard m^3/kg (0.2 to 0.75 std. ft^3/lb) of wet waste.

A number of operational techniques can be considered for potential enhancement of landfill gas generation. These techniques, applicable only to new landfills, are summarized in Table 25.

Costs of lining materials and of various operating methods also affect the economic feasibility of gas recovery schemes. Among the alternative materials for lining (bottom and sides), impervious soil, if available, is by far the lowest-cost material that would prevent leachate percolation into the groundwater table, whether or not gas recovery is included in the landfill development. If suitable low-permeability soil must be imported, the distance will determine the degree of competitiveness with other materials such as clay-treated on-

TABLE 24. MINIMUM LFG RECOVERY RATE REQUIRED FOR ECONOMIC VIABILITY[a]

LFG process/product	HHV kJ/std m^3	Private enterprise		Local government	
		Retail std m^3/min	Wholesale std m^3/min	Retail std m^3/min	Wholesale std m^3/min
Dehydration and compression	17727	33.4	NA[b]	17.7	NA
Upgrading (H_2O and CO_2 removal)	27990	NA	NF[c]	NA	47.2[d]
Upgrading (H_2O, CO_2, and N_2 removal)	36387	NA	NF	NA	141.5[d]
Upgrading and blending (H_2O and CO_2 removal with propane blending)	37320	NA	NF	NA	78.6[d]
Steam generation (using raw LFG)	17727	19.7	NA	9.8	NA
Electricity generation (steam turbine)	17727	35.4	NF	11.8	31.4
Electricity generation (gas turbine)	17727	19.7	45.2	9.8	15.7
Electricity generation (gas engine)	17727	45.2	NF	3.9	35.3
Methanol synthesis	17727	NF	NA	167.0[d]	NA

[a] Based on retail price of $2.00/1.05 MM kJ (1 MM Btu) for gas product, $2.56 for steam, $0.05/3600 kJ (kWh) for electricity and $0.092/liter ($0.35/gal) for methanol; wholesale price of $1.65/1.05 MM kJ (1 MM Btu) of product for gas, $0.092/liter ($0.03/kWh) for electricity; 10 percent net ROI for private enterprise and local government operations (30 percent gross ROI for private enterprise). Source: Lockman Report.

[b] NA, not applicable; NF, not economically feasible.

[c] Can be feasible in specific situations such as Mountain View, California, landfill.

[d] Break-even operation for local government.

TABLE 25. DESIGN AND OPERATIONS ALTERNATIVE FOR ENHANCING GENERATION OF LANDFILL GAS[a]

Alternative	Potential effectiveness	Cost range,[b,c]
Site lining		
Impervious soils	Effective if permeability low	0.25 to 0.42
Treated soils	Effective if permeability low	1.50 to 1.92
Film barrier	Effective with proper base preparation and cover protection	1.05 to 4.60
Asphalt cement with sealant	Effective if properly installed on firm base	3.00 to 4.05
Moisture control		
Water distribution below cover	Highly effective	0.21 to 0.42
Semipermeable soil with humectant	Effective if kept moist	0.03 to 0.04 (add 1.00/yr/m^2) operating cost)
Controlled uniform negative pressure below cover	Effective if carefully balanced by wells with half normal spacing	0.63 to 0.84 (additional cost for recovery subsystems)
Leachate collection and recycling	Effective: pH control may be required	0.13 to 0.29 plus cost of neutralizing chemicals
Operations		
Single water application (truck or hose/spray nozzle)	Effective initially; long term effects unknown	0.02 to 0.03
Waste shredding without daily cover (final cover only)	Effects unknown; may hasten initial LFG generation and should increase waste permeability	4.18 to 8.63 per ton received
Permeable daily cover	Increase ease of movement LFG within confines of landfill	No additional cost if suitable soil available
Sewage sludge seeding of waste	Effects unknown; may shorten LFG generation initiation time	0.25 to 0.42 (drop charge may exceed spreading costs)
Leachate recycling with pH control	Effective	0.13 to 0.29 plus cost of neutralizing chemicals

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[a] Reference 3.

[b] Costs estimated for 40-ha (100-ac) fill area landfill.

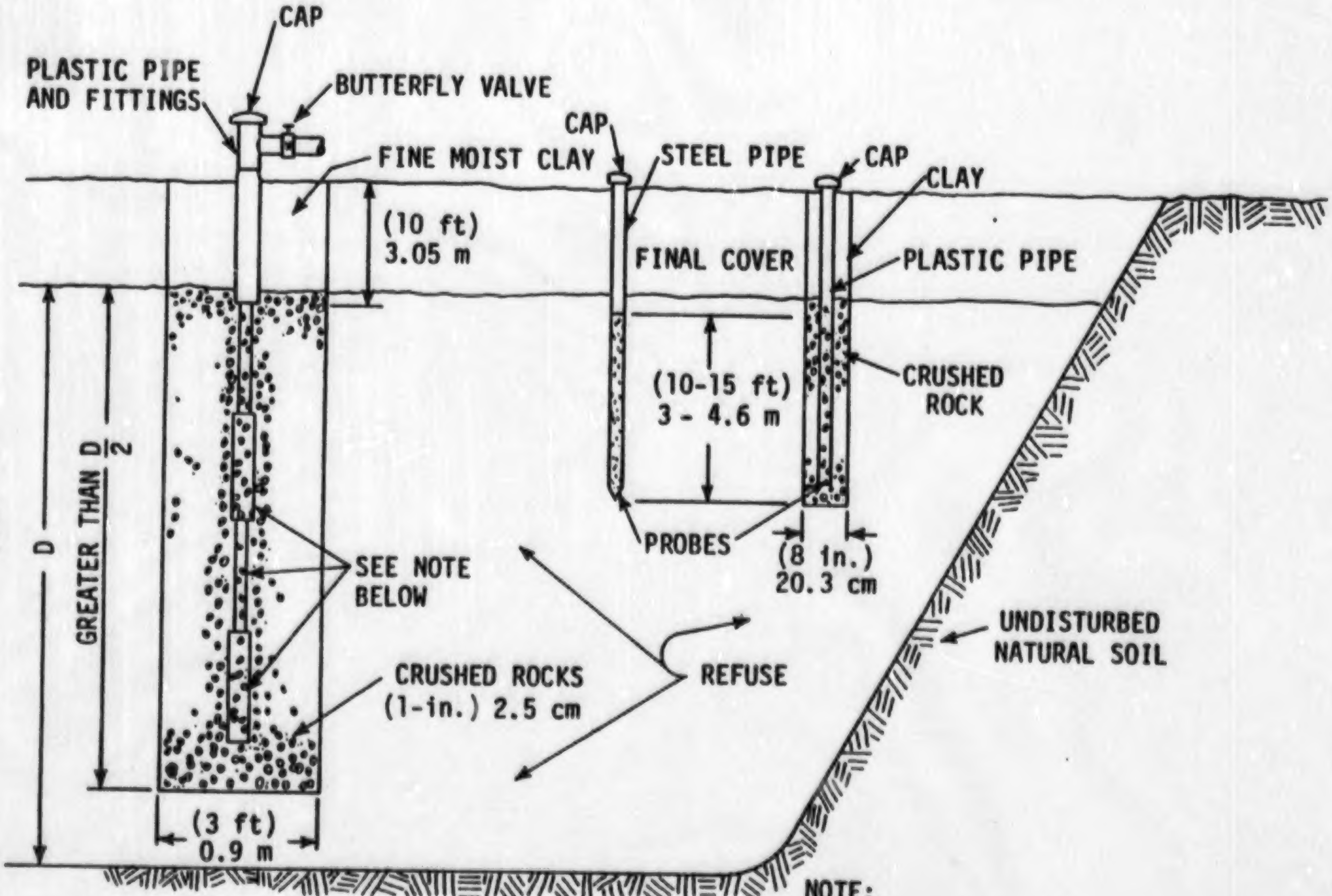
[c] 1 \$/m^2 = 0.09 \$/ft^2.

site soil. A reliable film barrier requires careful placement and can be several times more costly than impervious soil, as is also true of sealant-covered asphalt cement, which can be used only over a highly stable base.

Among moisture control techniques, collection and recycling of the leachate appears competitive with water distribution below the top cover. This technique is preferred if permitted by local water quality control regulations. Usually, where it can be demonstrated that the possibility of leachate seeping through the bottom lining is virtually negligible, collection and recycling is permitted.

Concerning the operating alternatives, little information is available with which to make evaluations or postulate probable effectiveness. If water loss with landfill gas is substantial, a one-time addition of water to the waste prior to daily cover may be inadequate over the long term. Shredding of waste prior to landfilling has been in contention, and the benefits are not likely to offset the high costs. Use of a permeable daily cover that will temporarily prevent vector attraction but permit passage of moisture and gas can be effective in reducing internal flow resistance, but not in preventing odors.

The effects of seeding solid waste with sewage sludge are not known with certainty and should be evaluated in controlled tests. Leachate recycling with pH control is both an operating technique and a design approach because it requires design and construction of special facilities. A typical landfill gas recovery well is depicted in Figure 27.


CAP
PLASTIC PIPE AND FITTINGS
BUTTERFLY VALVE
FINE MOIST CLAY
(10 ft) 3.05 m
CAP
STEEL PIPE
CAP
CLAY
FINAL COVER
PLASTIC PIPE
CRUSHED ROCK
(10-15 ft) 3 - 4.6 m
PROBES
(8 in.) 20.3 cm
D
GREATER THAN D/2
SEE NOTE BELOW
CRUSHED ROCKS (1-in.) 2.5 cm
REFUSE
UNDISTURBED NATURAL SOIL
(3 ft) 0.9 m

NOTE:

TYPICAL PERFORATIONS ON 6-in. (15.2 cm) PIPE - 8 SLOTS PER ROW 2 x 1/4-in. (5.1 cm x 0.6 cm) SLOTS ON 6-in. (15.2 cm) CENTERS, 4-in. (10.2 cm) PIPE - 6 SLOTS PER ROW

Figure 27. Typical landfill gas recovery well and probe.

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APPENDIX A

Examination of Table A-1, which lists reported values for leachate constituents, leads to the following observations. Leachate derived from recently deposited solid waste is generally acidic (pH 5.0 ± 1.0) and high in biologically oxidizable organics (BOD ≅10,000 mg/liter, COD ≅16,000 mg/liter, and TOC ≅5,700 mg/liter), and total solids (>1 percent). Concentrations of heavy metals are generally less than 1 mg/liter, but zinc concentrations often reach 10 to 100 mg/liter, and lead occasionally exceeds 1.0 mg/liter. Iron is commonly present at 1000 mg/liter. The pH of leachate derived from more stabilized solid waste generally is near 7 and the leachate is low in biologically oxidizable organics (BOD ≅100 mg/liter, COD ≅1000 mg/liter, and TOC ≅300 mg/liter) and total solids (2000 mg/liter).

The data in Reference 1 were obtained at the Boone County Field Site Test Cell 1, which contains 395 tons (wet) of municipal solid waste compacted to 592 kg/m^3. Test Cell 1 was constructed in 1971 in accordance with best sanitary landfill technology available at that time.

The data from the University of Illinois[2] were obtained under U.S. EPA Research Contract 68-02-0162. The leachate was generated from a laboratory lysimeter 1.22 m in diameter with a 38-mm grate opening, containing 1520 kg (wet) of shredded solid waste compacted to 330 kg/m^3 (wet). Water was applied daily to bring the solid waste to field capacity within 30 days, and thereafter an equivalent of 0.89 mm/week was added to generate sufficient leachate for evaluation of treatment methods.

Data reported by Drexel[3] were obtained under U.S. EPA Research Grant R800777. The solid waste was synthesized from selected waste streams to simulate residential solid waste; the laboratory column was 1.83 m square and initially was packed 2.44 m deep.

Data from the Georgia Institute of Technology[4] were obtained under U.S. EPA Research Grant R801397 to investigate the feasibility of recirculating leachate through the landfill as a treatment method. Again, the landfill conditions were simulated for study purposes. The refuse was compacted into a 3.05-m by 0.917-m column in two 1.52-m lifts to a dry density of about 318 kg/m^3. To expedite the production of leachate, 948

TABLE A-1. LEACHATE COMPOSITION[a]

Constituent	SHWRD[1]	U of Ill.[2]	Drexel[3]	Ga. Tech[4]	Ga. Tech[4] Recirculate	Olin Ave.[5] Landfill	Dupage[6] MW61	Dupage[6] MW63	Dupage[6] LW58	Dupage[6] LW68
COD	16,000-22,000	45,000-71,000	1,000-51,000	4,320-12,000	4,280-9,288	2,700-10,650	360	2,940	8,000	40
BOD$_5$	7,500-10,000			2,500-11,000	2,750-6,900	1,550-8,450	125[c]	4,560[c]	14,080[c]	225[c]
TOC	4,200-13,840	14,000-28,000		1,230-5,000	256-2,798					
pH	5.2-6.4	4.5-6.0	3.7-8.5	5.2-5.6	4.8-5.4	5.9-8.1			6.3	7.0
TS	10,000-14,000	34,000	0-42,000	2,442-12,500	1,627-6,918	4,028-7,790				
TDS	10,000-14,000	15,600-23,000					1,104	5,910	6,794	1,198
TSS	100-700	139	10-26,500	34-610	12-385					
Specific Conductance	6,000-9,000	9,400-16,800								
Alkalinity (CaCO$_3$)	800-4,000		0-9,700	558-2,280	302-1,370	1,800	1,630	4,720	5,810	2,250
Hardness (CaCO$_3$)	3,500-5,000		0-5,500	450-1,940	370-1,040	1,400	690[d]	2,250[d]	2,200[d]	540[d]
Total-P	25-35	20-100	0-130	2.8-26	0.65-22	12	0.5	0.17	1.2	8.9
Ortho-P	23-33	6.5								
NH$_4$-N	230-430	400-1,000	0-482	56-187	68-114					
NO$_3$+NO$_2$-N	0.2-0.8	0.5-10.0					0.14[e]	0.5[e]	0.7[e]	1.6[e]
Ca	900-1,700	2,300-4,000		125-750	60-435	80	156	447	308	102
Cl	600-800	1,500-2,500	4.7-2,340	90-385	91-248	400	205	946	1,330	135
Na	450-500	750-1,600	0-7,700	64-143	62-109		63	615	810	74
K	295-310	860-2,300					85	220	610	100
Sulfate	400-650	900-1,550	25-450	81-156	12-138		1	1	2	2
Mn	75-125			3-10	4-65	20	0.24	0.09	0.06	0.06
Mg	160-250	530-1,100		26-75	17-63		110	725	450	90
Fe	210-325	730-2,200	0-1,716	9-95	4-110	330	106	12	6.3	0.6
Zn	10-30	104	0-167	74	<0.05	5.5	0.10	0.05	0.4	4.5
Cu	0.01-0.60	0.5	0-9.9	0.48	<0.03		<0.5	<0.5	<0.5	<0.5
Cd	0.02-0.60			17	<0.03		BDL[e]	BDL	BDL	BDL
Pb	0.04-1.6	1.25		1.16	<0.10		1.0	1.0	0.5	1.0

TABLE A-1. (continued)

Winnetka[6] LW17	Winnetka[6] LW13	Winnetka[6] LW5B	Riverside[7] Bin #1
4,260	153	415	
6,400[c]	105[c]	250[c]	81-33,100
			5.6-7.6
2,306	584	894	
3,370	1,450	3,080	730-9,500
1,180[d]	710[d]	970[d]	650-8,120
2.6	1.3	2.75	0.2-29
0.5[e]	0.2[e]	0.43[e]	0.2-290
100	109	109	115-2,570
429	70	701	96-2,350
298	34	348	85-1,805
168	39	220	28-1,860
13.6	5	1	
1.14	0.2	0.1	
198	75	200	64-410
96	11	13.0	6.5-305
1.5	0.1	0.0	
<0.5	<0.5	<0.5	
BDL	BDL	BDL	
1.0	1.0	0.5	

TABLE A-1. (continued)

Constituent	Mission Canyon Landfill[8] 3-16-68	3-24-71	Sonoma, Calif.[9] Control	Recirculate	W. Va. U.[10] Cylinder C	Madison, Wisc.[2] milled uncovered	Madison, Wisc.[2] unmilled covered	Range of all values
COD	76,800	3,042	26,750-33,500	89,520		71,680	16,580	40-89,520
BOD_5	10,900	908	15,900-24,600	20,400	33,360			81-33,360
TOC						27,700	5,906	256-28,000
pH	5.75	7.4	4.3-5.4	4.6	5.88	5.97	5.72	3.7-8.5
TS					59,200	55,348	7,930	0-59,200
TDS	44,900	13,409	15,190-16,890	21,010				584-44,900
TSS	172	220	128-323	238		202	192	10-700
Specific Conductance						16,800	2,810	2,810-16,800
Alkalinity ($CaCO_3$)	9,860	8,677	0-5,480	3,050	20,850			0-20,850
Hardness ($CaCO_3$)	22,800	8,930			10,950			0-22,800
Total P	0.24	0.65	0.38-9.8	79.2	128	98	85	0-130
Ortho-P						29	85	6.5-85
NH_4-N	0	270	.33-304	194	1,106	1,028	347.4	0-1,106
NO_3+NO_2-N			3.2-4.3	4.7^e		10.29	4.29	0.2-10.29
Ca	7,200	216	1,041-1,700	1,560	2,790	3,900	572	50-7,200
Cl	660	2,355	1,200-1,300	1,210	2,310	2,467	474	4.7-2,467
Na	767	1,160	820	980	1,439	1,580	330	0-7,700

liters of tap water was added after placement of the soil cover. Pohland and Mao[4] provide a more detailed description of the project and the results to date.

The Olin Avenue data[5] are from the work done at the University of Wisconsin to determine the treatability of leachate by a variety of treatment methods. The landfill was constructed of shredded solid waste from the City of Madison.

The DuPage and Winnetka data[6] are taken from Hydrogeology of Solid Waste Disposal Sites in Northeastern Illinois, which specifically deals with leachate and groundwater quality in the vicinity of landfills.

Research conducted by Merz[7] at Riverside, California, represents the earliest extensive leachate study. The basic constituents identified in the test bin appear to agree with the most recent analysis of leachate.

The data from the Mission Canyon landfill[8] give a good indication of the effects of age. The early leachate analyses show high COD and BOD_5 values typical of fresh leachates. Some 3 years later the effects of age and materials are evidenced in the low COD and BOD_5 values; most readily oxidizable and biodegradable organics were removed. Significant increases in chlorides, sodium, and potassium over a 3-year period indicate that soluble material is available for leaching, perhaps in greater quantities because of the vigorous microbial activity within the landfill.

The Sonoma data[9] are from U.S. EPA Demonstration Grant 1-GOG-EC-00351, which evaluated the effect of high moisture throughout on leachate characteristics and refuse stabilization.

Laboratory-scale sanitary landfills containing a combination of shredded and unshredded solid waste were evaluated by Qasim and Burchinal.[10]

Chian et al.[2] reported on the characteristics of two leachate samples provided by R. K. Ham at the University of Wisconsin. The increased surface area of the shredded solid waste and lack of cover material drastically increase the quantities of materials leached.

A detailed characterization study by Chian and DeWalle[2] showed that 78 percent of the organic matter in fresh leachate consisted of low-molecular-weight compounds, 78 percent of which was free volatile fatty acids; also, significant amounts of the heavy metals were chelated by the humic carbohydrate-like large molecules and the fulvic acid small molecules. Leachate from an old landfill, consisted almost entirely of small molecules of fulvic acid and humiclike materials capable of chelating heavy metals; no free volatile fatty acids were detected.

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